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An Evaluation of Methane Mitigation Alternatives for Closed Municipal Landfills

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An Evaluation of Methane Mitigation Alternatives for Closed Municipal Landfills

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An Evaluation of Methane Mitigation Alternatives for Closed Municipal Landfills

by

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Countries around the world face social, economic, and ecological damage from escalating natural disasters caused by climate change. In an effort to curtail climate change impacts, local and regional governments are beginning to employ green house gas (GHG) mitigation strategies to reduce their carbon footprint. These strategies work to eliminate a range of GHG emissions from entering the atmosphere. Apart from carbon dioxide (CO_2), the most prevalent GHG is methane. In terms of global warming, methane is approximately 21 times more harmful to the atmosphere than CO_2. Natural gas systems, coal mining, manure management, rice cultivation, wastewater treatment, and landfills all contribute to methane generation. According to the US Environmental Protection Agency's 2011 US GHG inventory, landfills generate 1.5% of total GHG emissions in carbon dioxide equivalents. Recognizing the global impacts of its policies and operations, municipalities are working to reduce their GHG emissions. Coalitions like the C40 Cities Climate Leadership Group were created to specifically address GHG reductions, which will result in a 248 million MT reduction in GHGs released to the atmosphere by 2020.

Guided by existing literature, this Master's Report calculates methane generation and transport to determine the effectiveness of applying two methane

mitigation alternatives—passive methane oxidation biocovers (PMOBs) and landfill gas to energy technologies (LFGTE)—at an inactive landfill site to reduce GHG emissions. LFGTE generates energy for direct use such as space heating or industrial processes or for electricity generation. Cost-saving strategies abound for landfills which utilize LFGTE. PMOBs optimize the landfill surface soil cover environment to promote microbial growth of bacteria, called methanotrophs, which convert methane into carbon dioxide. When employed, these mitigation alternatives are designed to significantly reduce methane emissions from landfills.

The EPA has developed a computer modeling program (LANDGEM) to aid in the calculation of landfill gas generation. A hypothetical case study of a one million ton landfill was created and modeled for methane generation over a 35 year period. With methane generation rates calculated, assessment of potential LFGTE was performed and methane oxidation rate calculations were made to determine the impact of a PMOB and LFGTE on net GHG emissions at the landfill. The overall GHG reductions with these engineering controls were two-thirds of the level a landfill without controls would emit. These results indicate that implementing methane mitigation steps at closed landfills throughout the world would yield significant reductions in GHG emissions.

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Overview

Currently, countries around the world face social, economic, and ecological damage from escalating natural disasters caused by climate change. In a report from a leading global reinsurance intermediary, natural disasters worldwide caused $200 billion USD in economic damages in 2012 (AON Benfield, 2012). Of that total, United States insurers and the federal government paid $129 billion in domestic recovery (Lashof and Stevenson 2013). In an effort to curtail climate change impacts, local, regional, and national governments are beginning to employ green house gas (GHG) mitigation strategies to reduce their carbon footprint. These strategies, which include public policies and engineering controls, work to eliminate harmful GHG emissions from entering the atmosphere and altering climate patterns.

The most prevalent GHG, apart from carbon dioxide (CO_2), is methane, a gas which is approximately 21 times more harmful to the atmosphere in terms of climate change potential than CO_2 (Solomon et al. 2007). Natural gas systems, coal mining, manure management, rice cultivation, wastewater treatment, and landfills all contribute to anthropogenic methane generation. The third largest source of US anthropogenic methane emissions, which represents 17.5% of all methane emissions, is off-gassing caused by reactions in decomposing municipal solid waste (MSW) in landfills (EPA 2013c).

Guided by a recent literature review, this Masters Report will explain the current understanding of how methane is generated in a landfill and how it is calculated. Using site-specific information with a current EPA model, methane generation estimates will be calculated for a hypothetical inactive landfill. The report will also examine the current status of two mitigation strategies, landfill gas to energy technologies (LFGTEs) and passive methane oxidation biocovers (PMOBs). The net GHG emissions with mitigation

are then compared to emissions without the two controls. This will provide a resource for cities looking to quantify the benefits of LFGTE and PMOB at their municipal landfills.

Background

In the United States, Congressional inactivity on carbon emissions regulations has led to a lack of a comprehensive national mitigation policy and the presence of piecemeal executive orders and regional and local initiatives like the Regional Greenhouse Gas Initiative in the northeastern US and the California Cap and Trade Program (Carlson 2013). These executive branch and regional policies catalyze implementation of mitigation technologies from the bottom-up instead of the top-down to achieve GHG emission reductions (Larson 2013).

On the local level, cities can contribute to the climate change response by passing resolutions and ordinances to guide less carbon-intensive municipal operations. For example 2007, the City Council in Austin, Texas, passed a resolution,[1] which established a wide range of GHG mitigation policies for city operations, aided by a newly formed Climate Program within the Office of Sustainability. Section 1 of the resolution requires COA to:

> …make all COA facilities, fleets and operations totally carbon neutral by 2020 through measures including: …(c) developing and implementing departmental climate protection plans, including policies, procedures, targets, benchmarks and reporting for maximum achievable reduction of greenhouse gas emissions and energy consumption in all City departments.

Austin is also a member of the C40 Cities Climate Leadership Group and reports its GHG inventory to an international registry with third-party verification (COA, 2013). The C40, a coalition of 58 major cities, which represent 18% of the global GDP, across the world have agreed to a mission to reduce GHG emissions. Combined, the C40 have enacted over 5,000 measures relating to climate change which will curb emissions by 248 million tons of carbon dioxide equivalents (CO_2e) by 2020 and up to 1 billion tons CO_2e by 2030

[1] Resolution 20070215-023

(AP 2013). One such measure is converting one of the world's largest landfill located in Mexico City into an electricity generator. This move has the potential to reduce Mexico City's GHG emissions by 25 million metric tons (MT) CO_2e in 25 years (C40 Blog 2011).

In an attempt to reduce GHG emissions in the United States, the EPA enacted the New Source Performance Standards for landfills in 1996. This rule, which is derived from the Clean Air Act, requires landfills with a waste capacity greater than 2.5 million m^3 and which emit greater than 50 Mg $year^{-1}$ of nonmethane organic compounds (NMOCs) to install engineering controls to reduce LFG emissions (Morgan and Yang 2001). LFG emissions can pool from preferential pathways in landfills and create human and ecological health risks and can cause unintended explosions. Smaller, inactive landfills meeting the New Source Performance Standards criteria employ passive ventilation systems via a field of ventilation wells. Though the explosion risk is minimized, this alternative still poses human and ecological health concern, as around 550 NMVOCs and trace chemicals present in LFG are released with methane and CO_2 into the surrounding environment. Most sites, however, employ a LFG extraction system with a field of extraction wells and flare. A pump or blower is connected to the well field vacuums the LFG out of the soil. The flare then combusts the LFG. One disadvantage to this alternative is if the flare is not at optimal temperature, incomplete combustion occurs and unburned methane and NMOCs are released into the atmosphere. If the methane concentration is too low, the flare will expire, and a supplementary fuel source will be required to keep the flare ignited (Rajaram et al. 2012). The only other available or emerging mitigation alternatives the EPA identifies for methane reduction at landfills are LFGTE and biological remediation systems, like PMOBs (EPA 2011).

Instead of releasing LFG unabated into the environment or combusting it with a flare, LFG can be captured and transported to a methane-powered electricity generator or can be refined to high grade commercial or liquefied natural gas. The first LFG-powered electric generator began running in 1975, and now LFTGE alternatives are successfully

utilized by landfills across the world. In Texas, LFG is being converted to liquefied natural gas as an alternative fuel for vehicles (Zietsman et al. 2008). As of July 2013, over 600 US landfills generate energy. An additional 450 sites are candidates for LFGTE technologies (EPA 2013a). These sites are part of of 2,000 active and 3,200 inactive MSW landfill in the US (EPA 2013c). Current LFGTE sites employing LFGTE have the capacity to provide over 2 GW of electricity--enough electricity to power 2.8 million homes. US LFGTE systems reduce the amount of CO_2e released to the atmosphere by 103 million MT annually (EPA 2013a). In 2011, the US emitted 6,702.3 million MT of CO_2e, so LFGTE technologies have a noticeable impact on the US carbon footprint by reducing emissions by 1.5% (EPA 2013c). They displace the demand for more intensive GHG-emitting electricity generators with less intensive LFGTE alternatives. Cities like Los Angeles, California, have been successful in implementing these mitigation alternatives to reduce their carbon footprint and generate additional revenue. Methane generated from the largest landfill in the country outside Los Angeles fuels a 50 MW Rankine Cycle steam generation power plant which sells electricity back to the grid (Grenoble 2013).

Not all LFGTE alternatives are appropriate for all landfills. Each alternative has a range of methane flow rates with which it can operate. Calculating accurate present and future methane generation and emissions relies on a series of site-specific factors, which require field measurements before a municipality can report its landfills' methane emissions inventory or install a LFGTE. However, most methane generation models provide default values developed from empirical data, so general estimates of methane generation are possible.

The second mitigation alternative offered by the EPA is PMOBs. Through a biochemical process, PMOBs aid in the conversion of methane to CO_2. The cover, usually one meter in depth from the surface, contains constituents which promote a favorable environment for microbes to consume the methane and convert it into CO_2, a less potent GHG. An ex-situ, emerging technology also recommended by the EPA is

biofilters. Biofilters use the same bacteria to treat collected LFG by converting captured methane into carbon dioxide as the methane is pumped through a filter (Hettiarachchi et al. 2007). Researchers from Florida State University successfully performed pilot scale studies of PMOBs at Leon County, Florida landfill, which services the Tallahassee area. Although no field-scale PMOBs or biofilters exist (Abichou 2013), this seems to be a promising area for methane mitigation.

If feasible, these passive and active mitigation strategies could be used to limit the amount of CO_2e released from closed municipal landfills. About half of the landfill gas (LFG) generated from MSW decomposition in landfills is CO_2 and the remaining half is methane. The following section describes the biological and chemical reactions which yield this distribution.

Methane Generation in Landfills

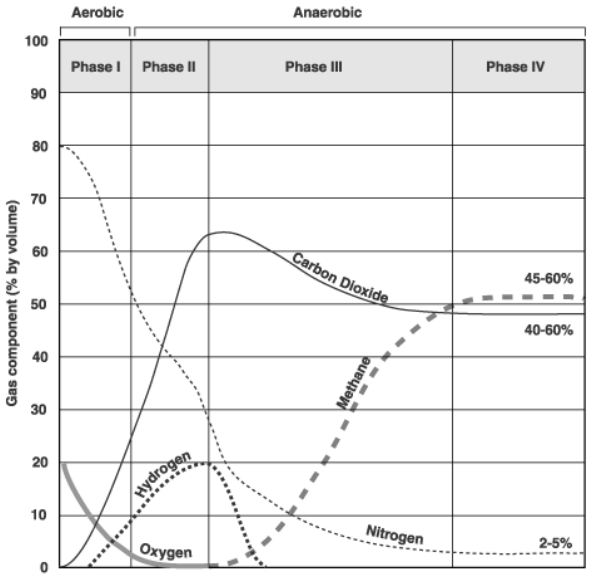
MSW biodegradation in municipal landfills is a multi-phase biochemical process, which takes decades to complete. Existing literature reports three to five stages for methane generation in a landfill, all with similar outcomes, waste biodegradation and LFG releases (Bove and Lunghi 2006; Themelis and Ulloa 2007; Rajaram et al. 2012; Abichou et al. 2006). Understanding the methane yields from a landfill is the first step in assessing what mitigation strategy is the best fit for the landfill. Methane yields are affected by MSW composition, the amount of MSW, the number of years the landfill accepted waste, etc. As previously mentioned, these values may sometimes be difficult to ascertain, so most methane generation calculations rely on default values from literature. Modeling programs, such as EPA's LANDGEM, aid in the estimation of future methane generation for strategic planning. LFG generation may range from 150 to 250 m^3 ton^{-1} wet MSW (Rajaram et al. 2012).

Chemical and Biological Processes

After MSW placement, the environment in the landfill cell evolves with the aid of naturally occurring bacteria from an aerobic, or oxygenated environment, to an anaerobic, or oxygen free, system. This transformation occurs over a period of months to years (Themelis and Ulloa 2006). The primary product of oxygen depletion through aerobic bacteria reactions is carbon dioxide. The type of MSW and initial oxygen concentration within the cell contribute to the speed at which the aerobic to anaerobic transformation occurs. Injecting supplemental oxygen into the cell or adding recycled landfill fluids, called leachate, will accelerate the decomposition process and increase the CO_2 and methane generation rates. Figure 1 summarizes the composition of LFG through the phases of biodegradation. Beginning in Phase III and maintaining through Phase IV, methane and CO_2 comprise 40-60% of the LFG by volume. This is the optimal period for LFGTE operations and PMOB activity. Phase V, not present in Figure 1, represents the maturation period when LFG concentrations achieve equilibrium with

the atmosphere, and methane and CO_2 diminish (Popov 2005). The following subsections explain in detail the four processes of waste biodegradation and methane generation.

Figure 1: LFG Gas Composition in a Closed Landfill



Source: ATSDR (2001)

Aerobic Process (Phase 1)

In Phase 1, daily landfill covers, meant to reduce odors and animals, cut off waste access to atmospheric oxygen. It is then that the first aerobic decomposition process begins producing CO_2 (Bove and Lunghi 2006). Abichou and Chanton (2009) reports that if bacterial oxygen demand is high enough, decomposition will begin before the cover is even placed. Bacteria naturally found in organic waste breakdown complex carbon chains that form the waste with the aid of oxygen and create water, heat, and CO_2 byproducts. The temperature in the landfill increases during this process because the reaction is exothermic. Phase 1 can take days to years depending on the level of compaction and type of waste accepted, which affect the initial oxygen concentration and oxygen demand, respectively (ATSDR 2001). The first aerobic reaction eliminating oxygen from the system (Robinson 1986) is given by:

$$6(CH_2O)_x+5O_2 \rightarrow (CH_2O)_x+5H_2O+5CO_2+heat \qquad \text{(Rxn 1)}$$

Acidogenic Process (Phase 2)

Once the oxygen supply is depleted and anaerobic conditions are established, anaerobic acid-generating bacteria proliferate in the waste. They react with the complex organic compounds to produce simple organic acids, such as acetic acid and ethanol. These acids can lower the pH to below 5. During the waste fermentation process with sugars and starches, heat generation and pH decrease as a result of increased production of acids and decreased oxygen supply. Hydrogen and carbon dioxide production continues. The following reaction shows glucose, available from the waste in the aerobic process, breaking down into ethanol and carbon dioxide.

$C_6H_{12}O_6 \rightarrow 2C_2H_5OH + 2CO_2$ (Rxn 2)

Acetogenic/Methanogenic Process (Phase 3)

Acetogenesis begins when available acids, which were generated by the acid-producing bacteria in Phase 2, oxidize and begin forming methane. Methanogenic bacteria, called methanogens, drive the oxidation reaction that creates methane and reduces the CO_2 percent-by-volume in LFG. The following reaction shows the dissolution of acetic acid to methane and CO_2.

$CH_3COOH \rightarrow CH_4 + CO_2$ (Rxn 3)

Methanogenic Process (Phase 4)

A second series of reactions in Phase 4, also catalyzed by methanogens, uses reactants available in the landfill cell and products from the acetate fermentation to produce more methane. CO_2 reduction with hydrogen in Reactions 4 and 5 explain the CO_2 decrease in percent by volume in LFG.

$2CH_2O$ (organic matter) + $2H_2O \rightarrow 2CO_2 + 4H_2$ (Rxn 4)

$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$ (Rxn 5)

Although most methane generation occurs within the first twenty years following landfill closure, methane can be emitted up to 50 years later (Popov 2005). Based on a study conducted by the US Air Force, the Air Force decided it unnecessary to remediate uncovered landfills older than twenty years, as they are unlikely to produce significant amounts of gas (Hauser 2009).

Calculating Landfill Gas Generation and Transport

Calculating methane generation from MSW biodegradation is the first step in determining the feasibility of LFGTE alternatives and the mitigation impact of PMOBs. Methane generation relies on obvious parameters such as total waste placement and less obvious parameters like nutrient availability for microbial activity. Fortunately, computer modeling programs like EPA's LANDGEM and published literature establish default values to aid in calculating a best guess for methane generation rates. As with all other models, the more site-specific data available, the more accurate the model becomes.

Conditions Impacting Methane Generation

Various conditions determine the rate at which a landfill generates methane and other LFGs. LANDGEM relies on a series of site-specific parameters such as the methane generation rate (k), the methane formation potential (L_o), the nonmethane organic carbon (NMOC); and the methane content to derive a source generation rate.

Methane Formation Potential (L_o): Each type of waste placed in a landfill has its own methane formation potential (m^3 CH_4 Mg^{-1} wet MSW), which is the quantity of methane produced per unit wet MSW over the landfill lifetime. Foodstuffs, sugars, starches, and other household wastes are highly biodegradable while moderately biodegradable waste like wood, paper, and garden waste produce less LFG. The MSW type with the least biodegradability is materials such as textiles. Waste that does not biodegrade, such as glass and metal, occupies the rest of the landfill space (Arigala et al. 1995). According to Ozturk et al. (2001), after final waste placement in a Turkish landfill, 46% of the waste

by weight is highly biodegradable, 19% is moderately biodegradable, 3% is least biodegradable, and 32% of the MSW does not biodegrade. Landfill-wide L_o values range from 6.2 to 270 m^3 CH_4 Mg^{-1} MSW. However, the default L_o value used by LANDGEM is 170 m^3 CH_4 Mg^{-1} wet MSW (EPA 1991). This value comes from New Source Performance Standards which guide methane control policies for US landfills.

Methane Generation Rate (*k*): The methane generation rate (yr^{-1}) is essentially the MSW decay rate. It is influenced by more variables than L_o, such as moisture content, nutrient availability for microbes, pH, and temperature (EPA 2005). Due to the complexity of *k*, it is calculated from empirical landfill data. Table 1 presents methane generation rate ranges reported in the literature. The New Source Performance Standards default value for *k* is 0.05 yr^{-1} (EPA 1991).

Table 1: Methane Generation Rates for Varying Waste Types

Site-Specific Condition	Generation Rate (yr^{-1})
Rapidly biodegradable waste	*0.14*-0.69[a]
Moderately biodegradable waste	*0.023*-0.14[a]
Slowly biodegradable waste	0.070-*0.017*[a]
Wet climate	0.1-0.35[b]
Medium moisture climate	0.05-0.15[b]
Dry climate	0.02-0.1[b]

[a]Based on data from *Arigala et al. (1995)*, Zanetti et al. (1997), and Lifshits et al. (1997); [b]Based on Landfill Control Technologies (1994); Source: Copty et al. (2004) and EPA (1996)

Nonmethane Organic Compound Concentration: The NMOC are trace chemicals which exist in LFG alongside CO_2 and methane. LANDGEM uses a default list of NMOCs with a total concentration of 4,000 ppmv representing 46 compounds (EPA 2005). Without site specific LFG sampling data, it is difficult to ascertain accurate trace chemical concentrations in LFG.

Methane Content: The methane content in LFG ranges from 40% to 60% by volume, and can be explained by stoichiometry and supported by empirical data. CO_2 occupies the remaining volume while other NMOCs and nitrogen occupy a very small percent of the LFG composition. Most calculations use a default value of 50% (EPA 2005).

Equations for methane generation calculation

There are various approaches to calculating methane generation at landfills from broad one-dimensional estimates to precise time-step calculations using multi-dimensional computer modeling programs (EPA 2005 and Arigala et al. 1995). The resources and data available dictate which approach best fits the methane generation calculation. This section provides tiered approaches to calculating methane source generation with the goal of producing the most accurate value.

Rule of Thumb Model

The broadest approach for calculating the methane generation rate is a rule of thumb ratio used by some LFG experts. The rate ratio ranges 0.003 m^3 to 0.012 m^3 of LFG generated per year per kg of MSW placed in the landfill. Due to its extremely broad approach, the model has a margin of error of +/- 50% (EPA 1996). Because methane occupies half of the LFG volume, the final value must be multiplied by 0.5 to determine methane generation. This ratio does not take into account local climate which affects *k* or varying MSW types which affect L_o, to name a few of the limitations of the model. As a result, this approach only provides a ballpark estimate for methane generation.

LANDGEM 3.02

EPA LANDGEM Version 3.02 is a user-friendly method to estimate annual LFG generation (Q_{source}). LANDGEM calculates Q_{source} (m^3 yr^{-1}) from a first order decomposition rate equation using the variables presented in the previous section. Version 3.02 differs from previous LANDGEM versions by incorporating shorter time steps for greater accuracy. The equation which expresses Q_{source} is given by:

$$Q_{source} = k * L_o * \sum_{i=1}^{n} \sum_{j=0.1}^{1} \left(\frac{M_i}{10}\right) * e^{-k*t_{ij}}$$

(Eqn 1)

where *k* is the methane generation rate (yr^{-1}); L_o is the methane formation potential by volume (m^3 CH_4 Mg^{-1} MSW); M_i is the mass of MSW accepted in the i^{th} year (Mg); *n* is

the year of calculation less the initial year of waste acceptance; *i* is one year time increments; *j* is 0.1 year time increments; and t_{ij} is the age of waste mass M_i (e.g., 3.2 years) (EPA 2005).

Instead of relying on New Source Performance Standards default values for L_o, Ritzkowski and Stegmann (2010) simplified a 1996 International Panel on Climate Change equation to more accurately calculate the methane formation potential by mass, or *L* (Mg CH_4 Mg MSW^{-1}). The Ritzkowski and Stegmann methane formation potential is given by:

$$L = MCF * F * \left(\frac{16}{12}\right) * DOC * DOC_F$$

(Eqn 2)

where *MCF* is a methane correction factor (*MCF*=1 for sanitary landfills; *MCF*=0.4-0.8 for waste dumps (Ritzowski and Stegmann 2010)); *DOC* is the fraction of biodegradable organic carbon by weight in the MSW (*DOC*=0.18-0.21 for North America (IPCC 1996)); DOC_F is the portion of the biodegradable organic carbon that degrades to LFG (DOC_F=0.5 default (IPCC 1996)); and *F* is the fraction of methane in LFG (*F*= 0.4-0.6 (Ritzowski and Stegmann 2010)). *MCF* takes into account waste placement history and whether the landfill was covered daily or left open to the air. The methane formation potential is reduced by the *MCF* if aerobic decomposition began before the landfill closed. *DOC* is determined by the waste composition of biodegradable MSW. Converting *L* to L_o requires the following conversion and uses the density of methane gas at 44°C[2]:

$$L_o = 1{,}620 * L$$

(Eqn 3)

[2] Average temperature of LFG samples collected from nine extraction wells at three different landfills (Barry 2008).

Findikakis and Leckie Model

Prior to LANDGEM, Findikakis and Leckie (1979) derived a gas generation equation with respect to time of waste placement for each landfill layer. The equation also accounts for the varying levels of waste biodegradation. The LFG generation rate, α (kg LFG m^{-3} MSW), is the sum of individual rates for component *i* and is given by:

$$\alpha = \sum_{i=1}^{n} \alpha_i(t) = C * A_i * \lambda_i * e^{-\lambda_i * t}$$

(Eqn 4)

where *C* is the total capacity for gas production (kg LFG m^{-3} MSW), which is comparable to L_o from Eqn 3; A_i is the fraction of the MSW corresponding with component *i* which will biodegrade into LFG; λ_i is the methane reaction rate constant (yr^{-1}) corresponding with component *i* (recommended to use Arigala et al. (1995) values from Table 1); and *t* is duration (yr^{-1}) from waste placement to desired period. For a methane generation rate, multiply α by the methane ratio in LFG of 0.5.

Because MSW biodegradation occurs in covered cells before the landfill is completely closed, Findikakis and Leckie (1979) accounted for the LFG generation disparity while the landfill is still accepting waste. Eqn 5 provides for *t* required in Eqn 4:

$$t = t_o + Z(\frac{t_f}{D})$$

(Eqn 5)

where t_o is time elapsed (yr) since the last layer of MSW was placed and the landfill was capped; t_f is the total time (yr) to fill the landfill; Z is the depth (m) of the MSW placement in the landfill; and D is the total landfill depth (m).

IMAGE Model

More recently, Staub et al. (2011) introduced a methane generation model, called IMAGE-Landfill model. Like the Findikakis and Leckie model, IMAGE accounts for time of waste placement; however, IMAGE also accounts for methane generation change in each cell. The result is a landfill-wide instantaneous methane generation rate (*Y(t)*) which can be used for resource planning. The first-order kinetic equation, known as the SWANA model, solves the rate as if the landfill was a single cell:

$$Y(t) = BMP * \frac{M}{1+w} * k * e^{-k*(t-\tau_{OB})} * \frac{k+s}{s} * (1 - e^{-s*(t-\tau_{OB})})$$

(Eqn 6)

where *Y(t)* is the methane generation rate (m^3 CH_4 ton^{-1} dry MSW); *BMP* is the biomethane potential (-m^3 CH_4 ton^{-1} dry MSW); *M* is the total MSW weight (tons wet MSW); *w* is the water content which converts wet to dry MSW (kg dry MSW kg^{-1} wet MSW); *k* is the methane generation rate (yr^{-1}); *t* is the time starting from the last disposal year to the time measured (yr); τ_{OB} is the relative time when methane production starts for a landfilled waste material (yr); and *s* is the shape parameter (yr^{-1}) (Staub et al. 2011). *BMP* is comparable to L_o. Based on a French landfill study, Staub et al. (2011) assumes τ_{OB} is 0.17 years. This delay is different from the previous models which assume generation begins before the landfill closes. The *k*, calculated from the waste biodegradation half-life value ($t_{1/2}$), is given by:

$$k = \ln(2) / (t_{1/2} - \tau_{OB})$$

(Eqn 7)

For calculating the incremental methane generation based on individual landfill cells, the first step is to calculate the waste quantity buried ($M_{cell(i)}$) in cell *i*. This is accomplished by multiplying the average humid density of the waste (ρ_h) by the volume

($V_{cell(i)}$) of cell *i*. Staub et al. (2011) uses 1.1 tones of wet matter m^{-3} of dry MSW. The time it takes to construct the cell ($t_{c(i)}$) in years is give by:

$$t_{c(i)} = M_{cell(i)}/m_i$$

(Eqn 8)

where m_i is the annual mass of waste buried (tons of wet MSW $year^{-1}$). The time delay for methane to begin forming from waste cell *i* is $\tau_{OB(i)}$. This time period accounts for both individual cell delay and landfill-wide delay in methane generation from the time of placement. $\tau_{OB(i)}$ is given by:

$$\tau_{OB(i)} = \frac{t_{c(i)}}{2} + \tau_{OB}$$

(Eqn 9)

Accounting for methane production in each cell, the following equation yields the IMAGE landfill-wide methane generation rate:

$$Y(T) = \sum_{m=1}^{i} BMP * \frac{M_{cell(i)}}{1+w} * k * e^{-k*(T-\tau_{OB(i)})} * \frac{k+s}{s} * (1 - e^{-s*(T-\tau_{OB(i)})})$$

(Eqn 10)

LFG transport through the landfill

Once the LFG is generated, transport mechanisms move it away from the source in every direction. Movement in some directions is more significant than in others based on site-specific conditions. Neglecting dispersion due to the significance of pressure build up in a landfill, the two LFG transport terms to consider are diffusion and advection (Stein et al. 2001). These forces balance out differences in concentration and pressure or move the gas via bulk motion, respectively. Darcy's Law explains the

advection while Fick's Law explains diffusion. Combined, the two laws form the general one-dimensional LFG flux equation[3] for a landfill which is given by:

$$J_i = -D_i * \nabla C_i + v * C_i$$

(Eqn 11)

where J_i= molar flux of gas component *i* (mol m^{-2} s^{-1}); D_i = diffusion coefficient of methane in soil (m^2 s^{-1}); ∇C_i = concentration gradient (mol m^{-2}); v = flow velocity of the gas mixture through the soil (m s^{-1}); and C_i = concentration of methane (mol m^{-3}). Flow rates are usually reported by mass, so to convert to mass, simply multiply the molar flux by the molecular weight of methane (16.04 g mol^{-1}). The diffusion coefficient in soil is less than the diffusion coefficient of methane in air because the methane travels through porous pathways. Diffusion of methane is soil is given by:

$$D_i^{soil} = D_i^{air} * \frac{a^2}{\phi^{\frac{2}{3}}}$$

(Eqn 12)

where D_i^{air} is the diffusion coefficient of component *i* in air (D_{CH4}^{air}= 2.275*10^{-5} m^2 s^{-1} (Hicks and Chopey 2012)); ϕ is soil porosity; and a is the volumetric air content. The flow velocity of the gas mixture through the waste or soil based on bulk motion is given by:

$$v = -\frac{k}{\mu} * \frac{\partial P}{\partial x}$$

(Eqn 13)

where μ is the gas mixture viscosity; *k* is the intrinsic permeability of soil; and *P* is pressure. Using the ideal gas law, pressure is calculated by:

[3] LFG travels through two medias before it reaches the surface, waste and soil. Most of the available research is on LFG transport through soil, so this report only examines that media's impact on LFG.

$$P = C * R * T$$

(Eqn 14)

where *C* is the molar concentration of methane (mol m^{-3}); *R* is the universal gas constant (8.314 J K^{-1} mol^{-1}); and *T* is the absolute temperature in the cell (Kelvins). The average temperature of LFG collected from seven wells at three landfills was 317 K (Barry 2008).

This completes the section on LFG generation and transport. Now that the necessary calculations and models have been presented to determine LFG generation, the next section presents available LFGTE alternatives which utilize this pseudo-renewable resource to off-set higher GHG emitting energy sources and reduce landfill operating costs.

Landfill Gas to Energy

LFGTE is a methane mitigation strategy, which recovers a significant volume of methane gas generated by landfills and converts it into electricity or other energy sources for direct onsite or near-site use. LFGTE offsets the need for non-renewable fossil fuels, such as coal, which would otherwise generate the same energy but with greater GHG emissions. LFGTE is a relatively inexpensive electricity source because the fuel is generated onsite and gas transmission distances are short.

According to the EPA's Landfill Methane Outreach Program, as of July 2013, active and inactive landfills across the US employ 767 LFGTE projects with an electricity generation capacity of 2,040 MW. This is a significant source of renewable energy, which represents 0.26% of the 2011 US summer load generating capacity (US EIA 2012). Local utilities and industry considering diversifying its energy supply portfolio should consider arranging agreements with local landfills to support their electricity needs, especially in times of peak demand and during periods of unpredictable electricity rates. In 2008, the Austin Community Landfill operated by Waste Management signed an agreement with Dell to provide 6 MW, or 40 percent, of their Round Rock, Texas, facility electricity needs from a LFGTE project (Waste Management 2013).

LFG is essentially a lesser grade of commercial natural gas. LFG has a methane composition of 40-60% while natural gas is 70-90% methane (naturalgas.org 2011). A benefit of LFG over commercial natural gas is that none of the emissions released from natural gas production are generated from landfill gas extraction systems, so the lifetime GHG emissions from LFGTE generation comes only from combustion. Onsite fuel supply for electricity generation also reduces emissions because transportation emissions are not created as compared to fuels like coal, which is brought by GHG-emitting rail to power plants. In terms of electricity generation, natural gas-fired systems and comparable LFGTE emit approximately 45 percent less carbon dioxide per KWh than the average US coal-fired power plants (US EIA 2013e).

Age and volume of waste placed factor into what LFGTE technologies are feasible. Closed landfills older than 50 years are usually in Phase 5 of the waste decomposition process and do not generate enough LFG to merit LFGTE.

In addition to CO_2 reduction, LFGTE reduces nitrogen and sulfur oxide (NOx and SOx) emissions generated from traditional coal-fired power plants. Nitrous oxide, N_2O, is a GHG coal combustion product, which is 310 times more potent than CO_2 in terms of global warming potential (ICBE 2000). Selection of natural gas-powered electricity generation over coal-fired reduces SOx emissions by four orders of magnitude (US EIA 1998) because natural gas has a sulfur content 3,200 times less than that of coal. As a result, the risk of short-term impacts of SOx exposure which causes bronchoconstriction and increases symptoms for asthmatics is reduced (EPA 2013b).

However, though LFGTE does reduce emissions, if a municipality seeking LFGTE is in a Clean Air Act non-attainment area for sulfur or nitrogen dioxide, the LFGTE alternative must go through a New Source Review. The review, a requirement of the Clean Air Act, forces a municipality to install the LFGTE alternative with the lowest achievable emissions rate, offset emissions elsewhere in the non-attainment area, and provide opportunity for public participation (EPA 2013d).

In addition to GHG emissions reductions, LFGTE technologies reduce landfill electricity and other energy related expenditures. In 2012, the Watauga County Landfill in North Carolina installed two 130 kW combined cycle generators powered by onsite LFG to meet the majority of its electricity demands. Between April and December, the landfill bought 70% less electricity from the local utility than it had historically, which resulted in a 48% decrease in electricity costs. In addition, the landfill sold $17,800 of electricity back to the utility during that time period at 5.53¢ per kWh (Hoyle 2013). This is a significant cost savings, which can be passed down to the consumer.

Landfill Gas Extraction System

Installing a LFG extraction system is the next step in harnessing energy from methane. Once a site assessment is performed and a methane generation rate is calculated, the methane capture efficiency rate from the LFG extraction system, or proposed LFG extraction system, should be determined. This efficiency rate is the percent of total LFG generated that is captured by the LFG extraction system. Table 2 presents LFG extraction system capture factions based on cover type. Once the efficiency rate is known, a range of LFGTE alternatives becomes feasible based on the available LFG flow rate.

LFG extraction system wells are installed in a field equidistant from each other. The wells are screened at landfill depth where the maximum off-gassing occurs. This is usually occurs at 75% of the cell's depth (Rajaram et al. 2012). With the extraction wells installed, a pump vacuums the LFG from the cell. The effectiveness of a LFG extraction system relies partially on the type and quality of landfill cover, as summarized in Table 2. The less penetrable the cover is, the higher the LFG extraction system efficiency (EPA 2011). Condensate accompanies the LFG, so knock out valves are installed to reduce the saturation content. Further LFG treatment, like bag filters, is needed to remove contaminants, including siloxanes and particulate matter, before the LFG may be used for LFGTE fuel.

Table 2: Extraction System Capture Efficiencies

Scenario	*Efficiency (%)*
No extraction system	0[a]
Extraction system at an active site with daily soil cover	35[a]; 60[b]
Extraction system with intermediate cover	65[a]; 75[b]
Extraction system with final clay cover	85[a]
Extraction system with final geomembrane cover	90[a]; 95[b]

[a]Based on Spokas et al. (2005); [b]Based on 40 CFR 98, Subpart HH, Table HH-3; Source: EPA (2011)

LFGTE Alternatives

Once the extraction system captures the LFG, available technologies offer an array of LFGTE alternatives to harness LFG energy, ranging from electricity generation to local direct use. Alternative selection depends upon a variety of site-specific economic and environmental factors including capital costs, electricity demand, LFG flow rates and LFG quality.

Because the focus of this study is mitigation of GHG emissions, a common value, called GHG reduction factor, was derived for each alternative to compare their GHG reduction effectiveness. A higher factor means less CO_2e are emitted per unit of LFG used by the alternative. This factor was calculated by dividing the total tones of CO_2e emissions prevented per year for each LFGTE alternative reported to the EPA by the volume of LFG used per day by the alternative. Table 3 presents LFGTEs employed by landfills across the country and their contribution to GHG reductions quantified by their GHG reduction factor. For LFGTE electricity generation and direct use projects currently in operation, cogeneration provides the highest emissions reduction per unit of LFG used.

Table 3: Greenhouse Gas Reduction Impacts by LFGTE Technology

LFGTE *Local Direct Use* Alternative		LFGTE *Electricity Generation* Alternative	
Technology	***GHG reduction factor[a]***	***Technology***	***GHG reduction factor[a]***
Alternative Fuel	85.7	Cogeneration	41.4-140.8
Boiler/Direct Thermal	82.2	Combined Cycle	37.4-110.0
Greenhouse	82.2	Gas Turbine	50.9-111.5
High/Medium Btu	82.2	Microturbine	47.4-77.0
Liquefied Natural Gas	85.7	Organic Rankine Cycle	NA
Leachate Evaporation Systems	82.2	Internal Combustion Engine	11.5-232.3
Cogeneration	41.4-140.8	Steam Turbine	24.8-126.7
		Stirling Cycle	53.8

[a]Tons of CO_2e emissions prevented per year / one million ft^3 LFG used per day; Source: EPA (2013c)

Cogeneration/Combined Heat and Power and Combined Cycle

Cogeneration, or CHP, and combined cycle LFGTE technologies provide a more efficient approach to harness the maximum available energy released from LFG combustion for

local direct use or added electricity generation. Most LFG-powered generators run at 25% efficiency. Cogeneration and combined cycle offer significant increases in energy efficiency resulting in energy cost savings and reduction of unnecessary combustion and subsequent GHG emissions.

Waste heat from LFG-powered engines may be used for onsite heating or commercial/industrial processes. Heat exchangers on the engine capture the waste heat from heated exhaust or working fluids and transfer them to adjoining heat recovery systems. In Onalaska, Wisconsin, a regional healthcare system relies on 12% of its energy demands from a cogeneration LFGTE system. According to the DOE, hospitals are one of the most energy intensive buildings (Haefke and Zarecki 2011). Captured LFG is piped from the La Crosse County Landfill to one of the regional healthcare system's clinics where it powers a 1.1 MW internal combustion engine to provide some electricity for the hospital (Nichols et al. 2013) and the waste heat is harnessed for local use. Cogeneration alone has allowed the healthcare system to eliminate natural gas-fired boiler and hot water heater use for most of the year. This cogeneration brings the healthcare system one step closer to its goal of energy independence by 2014 (Burns & McDonnell 2012).

The 1.1 MW internal combustion engine's intercooler, lube oil, and jacket water generate waste heat during the combustion process. A decoupling heat exchanger transfers 2 million Btu hr^{-1} from the engine fluids to a heat recovery loop operated by the hospital. Heat exhaust is also captured from combustion and exchanged into 2 million Btu hr^{-1} at the loop for commercial use. The loop pumps 200^{o}F water to the hospital floors where it is used for space heating and hot water heating. This design increases LFG energy efficiency to 56%--up from around 33% for internal combustion engines alone (Nichols et al. 2013).

Most landfills generate waste heat from internal combustion engines, gas turbines, or microturbines. Some landfills use boilers or steam turbines which release heat to power an adjoining bottoming cycle, like a steam turbine or Organic Rankine Cycle (ORC) (EPA 2010). By capturing the waste heat and converting it to more electricity in a combined cycle design, the LFG energy efficiency is increased to 60% (Staub et al. 2011). For example at Greece's largest landfill, Ano Liosia, 15 internal combustion engines with a capacity of 23.5 MW convert 50% of LFG potential energy to electricity. This is due to the energy loss from waste heat released during the internal combustion engine electricity generation process. Gewald et al. (2012) performed a comparative analysis on the efficiencies of electricity generation from waste heat between a steam turbine and an ORC. In both designs, waste heat, which was captured from the internal combustion engines, increased the site electrical system efficiency and subsequently, capacity. The steam turbine contributed to a 4.8% rise in efficiency for an additional 2.8 MW of generating capacity (Gewald et al. 2012).

Local Direct Use

Local direct use of LFG energy is a proven strategy some landfills employ to reduce overhead operating costs. LFG is used as an alternative to commercially or utility-supplied natural gas, propane, etc. Site-specific conditions should be taken into consideration when choosing a LFGTE alternative like proximity to commercial building or industrial process for heating applications. However, local direct use is not specifically for onsite use. Nearby business can use LFG as an alternative fuel source, as well. Heat is the most common LFG local direct use alternative; however, treated LFG can be converted to liquefied natural gas for modified vehicles. LFG is also used to fire boilers or infrared tube heaters for heating large open spaces in buildings.

Boilers and Direct Thermal

Today, US landfill boilers combust up to 6.48 million ft^3 of LFG per day (EPA 2013a). Heating boilers by LFG for heat transfer purposes is an attractive LFGTE alternative

because boilers require relatively low-grade gas to operate. Once the LFG is captured, it is sent to a boiler where a flame ignites the gas and transfers heat to another media, usually water for steam. The heated media is used for processes requiring high temperatures or for space heating. The energy yield from LFG combustion is not as high as natural gas combustion. This is due to LFG impurities and a lower percentage of methane in LFG. It takes twice as much to LFG generate the same amount of energy as with natural gas (Rajaram et al. 2012).

Innovative LFGTE technologies can be applied for direct thermal use because most industrial processes require heat. Industries such as cement or brick kilns, asphalt hot mix plants, glass furnaces, and incinerators use direct thermal heating from LFG in their operations. However, transport costs should be considered unless the industry is in close proximity the landfill, as the cost to procure LFG competes with the market price of natural gas. With the recent oil shale boom opening up large reserves of natural gas, natural gas has flooded the market. The industrial price of a cubic foot of natural gas has decreased from $6.76 in 2007 to $3.02 in 2012 (US EIA 2013a). It is unlikely future industrial designs will incorporate direct thermal from LFG until natural gas returns to pre-boom prices. The Department of Energy estimates that natural gas prices will rise again around 2035 (US EIA 2013b).

To reduce overhead costs, some landfill managers have transitioned to infrared tube heating systems for their facilities. Infrared heating occurs when combustible gases, like LFG, are mixed with air and ignited in a combustion chamber. The exhaust gases are blown through a heat exchanger tube and emit infrared waves. These waves generate thermal energy, which heats large areas (Detroit Radiant Products 2013). According to Dudkiewicz and Jeżowiecki (2009), thermal comfort can be maintained using infrared heaters at work stations while lowering the HVAC air temperature by 5^{o}C. When this is done energy consumption is reduced by 30%. The Fredrick County and I-95 landfills in Virginia invested in infrared tube heating for their storage and maintenance buildings

beginning in the early 2000s. The landfills recuperated their capital costs after 5 to 10 years which saved them $12,000-$15,000 per year in propane heating costs (SCS Engineers 2006).

Greenhouses

Infrared heating may also be used to heat greenhouses during the winter to simulate year-round growing seasons. The controlled temperature environment also allows growers to reduce growing time for variety of plant species with varying temperature demands. Once again, proximity to a landfill is a consideration with this technology. LFG-sourced heating for greenhouses is uncommon due to the stigma of growing edible plants near a landfill. However when employed, significant heating savings and production efficiencies are realized. Jaeger Greenhouses in Maryland Heights, Missouri, uses waste heat from the gas turbines which generate 14 MW of electricity at the Maryland Heights Renewable Energy Center. Jaegar Greenhouses experiences savings of $12,000-$15,000 per year in fuel related costs (Mulhem 2010).

Commercial Grade Gas (High and Medium Btu)

In 1975, the Palos Verde Landfill in California collected LFG, converted it to high Btu natural gas which was then sold to a local natural gas supplier. This was the first time LFG was captured and put to beneficial use. Since then at landfills around the world, low-grade LFG (less than 400 Btu ft^{-3}) has been treated and converted to medium- (between 400 and 600 Btu ft^{-3}) and high- (greater than 600 Btu ft^{-3}) grade gas stock (Morgan and Yang 2001). However, according a *Hydrocarbon Processing* Special Report, for energy companies to now accept commercial grade natural gas from landfills, the LFG must have a heating value greater than 900 Btu ft^{-3} (Nagl 2007).

In order to achieve this high heating value from low-grade LFG, an extensive treatment system must be installed. The treatment objective is to remove as many LFG impurities to increase the methane percent by volume. LFG impurities include hydrogen sulfide (H_2S), CO_2, NMOCs, chlorinated and fluorinated hydrocarbons, siloxanes, nitrogen,

oxygen, water vapor and particulate matter. Permeable membranes act as filters and allow CO_2 to pass through but not methane, molecular sieves separate oxygen and nitrogen out of the stock, liquid scrubbers remove H_2S, chemical adsorption takes care of water vapor, and activated carbon is the professional standard for hydrocarbon and siloxane treatment (Nagl 2007). An added benefit to high- and medium-grade gas is the LFG purification process removes the sulfur, so no sulfur is emitted during combustion, unlike sulfur-intensive coal combustion.

All technologies are expensive, yet viable, treatment options, which when combined, create a reliable stream of high-grade LFG. With the cost of commercial natural gas at historic lows, it is difficult for LFG to be competitive when capital costs for treatment are high.

Natural gas suppliers guarantee a certain heating value to their customers, so when the LFG heating value is less than the guaranteed value and blending is not an option, medium- grade LFG is used for other direct use commercial applications. Commercial examples include alternative fuels, which are discussed in more detail in the following subsection, boilers, or internal combustion engines (Morgan and Yang 2001). Table 4 presents the LFG heating values for some LFGTE alternatives.

Table 4: Required LFG Grades for LFGTE Alternatives

	LFG Quality Required		
LFGTE Alternative	*Low (<400 Btu/ft^3)*	*Medium (400-600 Btu ft^{-3})*	*High (>600 Btu ft^{-3})*
Internal Combustion Engine		X	
Gas Turbine	X		
Microturbine	X		
Steam Turbine	X		
Organic Rankine Cycle Engine	X		
Stirling Cycle Engine	X	X	X
Commercial Natural Gas			X
Alternative fuels		X	X

Source: Rajaram et al. (2012)

Alternative Fuels: Liquefied and Compressed Natural Gas

Two fuel alternatives to gasoline and diesel for vehicle engines are liquefied and compressed natural gas (LNG and CNG). Once purified and liquefied or compressed, LFG can be used to fuel waste haul trucks serving landfills. Waste haul trucks make tens of trips daily from their stop-and-go collection routes to the landfill. As a result, its fuel efficiency averages 2 mpg (Argonne National Laboratory 2012). Most waste haul trucks are diesel-powered and as of September 30, 2013, the cost of a gallon of US Gulf Coast Ultra-Low-Sulfur No. 2 diesel fuel was $2.93 (US EIA 2013c). The volatile cost of diesel and significant GHG emissions from fuel production and exhaust from these low MPG engines makes waste haul trucks a leading candidate for LNG or CNG. According a 2008 study from the Texas Transportation Institute, refuse trucks have a seven year life span making retrofits to new fleets fairly immediate (Zietsman et al. 2008).

The study also reports Mack Trucks, Inc. and Acrion Technologies developed a LFG purification process which separates methane from the NMOCs, CO_2 and water molecules. First the LFG is pretreated, likely through a bag house where a bag filter removes particulate matter. The LFG is compressed and hydrogen sulfur is removed before it is sent to a dryer to remove water molecules. In the three-story wash chamber

LFG is fed to the bottom of a column where a CO_2 liquid scrubber promotes NMOC separation. Refrigeration coils at the top of the chamber condense the majority of the CO_2 to pure liquid where it is captured and used for dry ice or food processes. The CO_2 may also be reused for scrubbing fluid (NETL 2001). The remaining LFG is now medium-grade gas for vehicles like waste haul trucks (Zietsman et al. 2008).

Argonne National Laboratories (2012) has developed the Greenhouse gases, Regulated Emissions, and Energy use in Transportation (GREET) fuel-cycle model for transportation planning to help cities determine the fuel mix for its service vehicle fleet. Municipalities could use this model to optimize their fleets, if they have not done so already. This report only uses the reference data from the model to show the emissions benefits of LNG and CNG over diesel. According to GREET, one waste haul truck, which travels 23,400 miles per year will emit 107.6 MT CO_2e running on diesel while a truck on LNG or CNG will emit 18.8 or 24.1 MT CO_2e, respectively. Although the three fuels emit similar CO_2e mass during combustion, CNG and LNG are credited CO_2e in the model for its fuel production CO_2e emissions savings.

Leachate Evaporation System

Leachate evaporation is a direct land use alternative which reduces the volume of leachate generated by the landfill. Along with LFG management, leachate management prevents environmental releases, which can cause groundwater contamination. Most large landfills install a leachate collection system with a series of leachate extraction wells screened at the deepest layer of a landfill. One leachate management alternative to costly direct disposal is to pump the leachate back through the cell, which increases the waste decomposition and subsequently increases the methane generation rate (Rajaram et al. 2012).

Traditionally, leachate evaporation systems (LES) are sludge fields or evaporation ponds, but these methods are time consuming. An expedited leachate evaporation alternative is to heat the LFG to 180-190 ^{o}F and run it through a leachate boiler. Heat from the

boiler vaporizes the water and combusts the VOCs before it is released into the atmosphere as CO_2. The concentration of solids in the leachate can reach thirty percent after vapor stripping. Omni-Gen Technologies, Inc. produces a LES which treats 10,000 gpd of leachate, but most are smaller in scale (Roe et al. 1998). This process significantly reduces the off-site disposal volume and costs and keeps harmful GHGs from entering the atmosphere. One million cubic feet per day of oxidized stripped leachate gas saves 80,000 tons of CO_2e. Thirteen field scale LESs treat leachate from 47 billion tons of waste across the US (EPA 2013a).

Electricity Generation

The overwhelming majority of LFGTE projects in the US generate electricity. Although LFG is not a true renewable energy, municipal waste generation will not cease until a zero waste world is achieved—unlikely in our lifetime. However, most in the energy field construe it as a renewable energy source. With the aid of LFGTE technologies, one million tons of municipal solid waste is capable of generating 0.8 MW of electricity (Rajaram et al. 2012). Apart from the 123 million ton-Puente Hills landfill which services Los Angeles, California, US landfills powering LFGTE electricity on average contain over 8 million tons of waste in place. Using that estimate with US EIA annual electricity use data, the average landfill could supply electricity demands for slightly fewer than 5,000 homes (US EIA 2013d). That means 5,000 less homes would potentially be relying on coal-fired electricity per landfill utilizing LFGTE technologies.

An Italian study compared life cycle GHG emissions of a one million ton-landfill without emissions controls to a landfill retrofitted with an internal combustion engine LFGTE. The study concluded the internal combustion engine's use of LFG over traditional electricity generation sources significantly reduces the amount of CO_2e released into the atmosphere. The LFG extraction system captured 80% of the LFG generated. The net amount of CO_2e emitted from the internal combustion engine was three times less than

that of the landfill with no controls (Lombardi et al. 2006). Table 5 summarizes the advantages and disadvantages for each LFGTE system.

Table 5: LFGTE Technologies Advantages and Disadvantages

LFGTE Electricity-Generating Technology	*Advantages*	*Disadvantages*
Cogeneration (CHP)	More efficient energy source resulting in energy cost savings[a]	NOx emissions[c]
Combined Cycle	More efficient energy source resulting in energy cost savings[a]	None
Gas Turbine	Lower GHG emissions than ICE; sufficient waste heat for CHP or combined cycle use[d]	Not as efficient as ICE; vulnerable to siloxanes fouling operations[d]
Microturbine	Small scale so works well with low flows; modular applications available[d]	Not as efficient as ICE; LFG needs pretreatment to remove siloxanes and moisture[d]
Organic Rankine Cycle Engine	Low mechanical stress; runs on low heating values; water demineralization unnecessary[b]	Not very efficient compared to other LFGTE technologies[f]
Internal Combustion Engine (ICE)	Short payback time[e]; high efficiencies[d]	High GHG emissions; low waste heat temperatures so limited CHP or combined cycle applications[f]
Steam Turbine	High efficiencies; low cost; not impacted by LFG impurities; works well on large scale applications[d]	High capital costs
Stirling Cycle Engine	Low GHG emissions[e]; relatively quiet during operation[h]	Negative payback to long payback time; small scale development only[e]

Source: [a]Nichols et al. (2013); [b]Turboden (2013); [c]Morgan and Yang (2001); [d]Rajaram et al. (2012); [e]Bove and Lunghi (2006); [f]Gewald et al. (2012); [g]EPA (2008); [h]Roe et al. (1998)

Capital cost for LFGTEs is becoming competitive. Gas turbine and small scale electricity generation technologies, like microturbines, are closing the price gap with traditional fossil fuel-supplied electricity generators. The US Energy Information Agency predicts that electricity produced from a conventional combined cycle natural gas-fired system will cost 33% less than a conventional coal system in 2018. The price disparity is even higher when pollution abatement technologies are introduced (US EIA 2013f).

Internal Combustion Engine

Internal combustion engines are the types of gasoline engines found in cars. Its design can also be applied to landfill settings where the LFG, not gasoline, is the primary combustion source. In an internal combustion engine, air is mixed at an optimal combustion ratio with LFG in a piston chamber. A spark plug ignites the compressed fuel mixture which forces the piston downward in the chamber. The piston is attached to a crank which turns a crankshaft. The crankshaft can either turn gears in a car or an electric generator at a LFGTE site (Arcoumanis 1988). The internal combustion engine's electricity generating capacity currently in operation ranges from 100 kW to 14.9 MW (EPA 2013a). Unfortunately with its many moving parts, internal combustion engine operation results in high maintenance costs.

Internal combustion engines are the most common and versatile electricity generation systems at landfills because they have low initial capital costs and can generate electricity with variable LFG flows at a fairly efficient rate. Like a car engine, its air/fuel ratio is controlled by computers to maximize the energy released from combustion. However, LFG combustion in internal combustion engines is not as complete as gas turbines; therefore, exhaust emissions containing GHGs and other air pollutants are higher (Rajaram et al. 2012). Gas turbines introduce surplus outside air into the ignition chamber which makes for a more complete combustion with the fuel than internal combustion engines (Kohn et al. 2011). In one combustion emission study, researchers at Columbia University confirmed this assertion that turbines combust fuel more completely than internal combustion engines. The study found turbine engines emitted significantly less CO, unburned hydrocarbons (UHCs), and NOx than a comparable internal combustion engine (Westby, 2008).

Gas Turbine

Second to the internal combustion engine, the gas turbine is the most used LFGTE technology at US landfills (Bove and Lunghi 2006). Recommended LFG flow for a 3 to 4

MW gas turbine is 1,400 ft^3 min^{-1} (EPA 1996). Despite their inefficiency, gas turbines emit fewer pollutants per kWh produced than internal combustion engines because, as previously stated, they introduce outside air during combustion which results in a more complete burn. Both technologies have a lifespan of over 25 years (Jaramillo and Matthews 2005).

At the most basic level, gas turbines are divided into compressor, combustion, and turbine modules. Outside air is brought into the system with a compressor through an air intake. The compressed air reaches temperatures of 850 ^oF and is then mixed with the LFG for ignition in a combustion chamber. The exhaust, released from combustion, is forced through a set of turbines which rotate a crankshaft on an electric generator. The exhaust leaves the system at around 1,000 ^oF, so most gas turbine designs capture the waste heat and reuse it to heat other processes or use it as supply heat for more electricity generation. The largest gas turbine generator, though not used in a LFTGE application, can generate 200 MW of electricity (Soares 2008).

The second largest LFG gas turbine electricity generation in the country is at the Maryland Heights Renewable Energy Center (Maryland Heights) outside St. Louis, Missouri. Maryland Heights, which was brought online in 2012, is the result of Missouri's energy diversification policy which is similar to Texas' Renewable Portfolio Standards. Maryland Heights is powered by 6,000 ft^3 min^{-1} of LFG from the Maryland Heights County Landfill which receives the majority of its waste from St. Louis. Once treated for siloxanes, H_2S and other UHCs, the LFG powers a series of three 4.8 MW Mercury 50 gas turbines which deliver electricity to the Ameren Missouri Distribution grid. Because the site is near a residential area, the system could make noise no greater than 50 decibels, so engineering controls, like enclosures and turbine silencers, were built into the design (Wibbenmeyer 2013).

Treatment for siloxanes is necessary because when LFG is not completely combusted siloxane leaves hardened silica deposits on metal surfaces and fouls the system (Galowitz 2013). Siloxanes are used in cosmetics, deodorant, water repelling coatings, food additives, soaps, lotions, and plastics (Wibbenmeyer 2013). One technology for siloxane removal, which is used at the Maryland Heights Renewable Energy Center, is a treatment system which compresses and chills LFG from 60-110 ^{o}F to 35 to -20 ^{o}F. The temperature and pressure changes cause siloxanes and moisture to separate out of gas phase (Rajaram et al. 2012). However, this treatment technology does not come without energy penalties and additional costs.

Microturbine

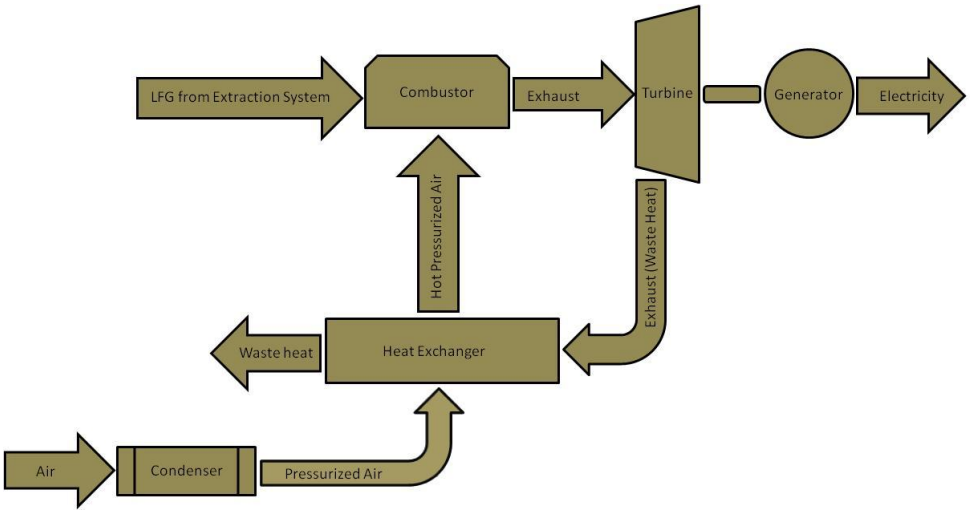
Microturbines generate electricity under a variety of site conditions. Because of a small layout, microturbines can process low flow rates near 350 ft^3 min^{-1} (EPA 2011). For larger sites with higher flow rates, microturbines can run in parallel for an increased electricity output. In the Village of Antioch, Illinois, the HOD Landfill supplies electricity to a nearby high school with microturbine electricity generation. The landfill is fitted with 12 Capstone MicroTurbines^{TM} which require 12 to 16 ft^3 min^{-1} LFG each and produce in total 360 kW for a nearby school. The system also includes cogeneration design where exhaust from the microturbines is transferred to the high school's boiler room for space heating needs (Rajaram et al. 2012).

Another benefit to microturbines is that they can be constructed for modular use. Microturbine modularization allows for ease of transport from one landfill cell to the next (Rajaram et al. 2012).

Figure 2 is a schematic of the microturbine process. Outside air is brought into a compressor where it is pressurized. With pseudo-cogeneration design, the air is then heated by exhaust heat from the turbine in a heat exchanger before it enters the combustion chamber with the LFG. This step increases the electrical efficiency of the turbine. LFG is then ignited with the heated, pressurized air in the combustion chamber

which forces exhaust to rotate the turbine blades to generate electricity. The combustion exhaust is sent to the heat exchanger to heat compressed air under pseudo-cogeneration design. The remaining waste heat may be used for further cogeneration [Combined Heat and Power (CHP)] or combined cycle applications as in the HOD Landfill design (Rajaram et al. 2012).

Figure 2: Microturbine Schematic with CHP

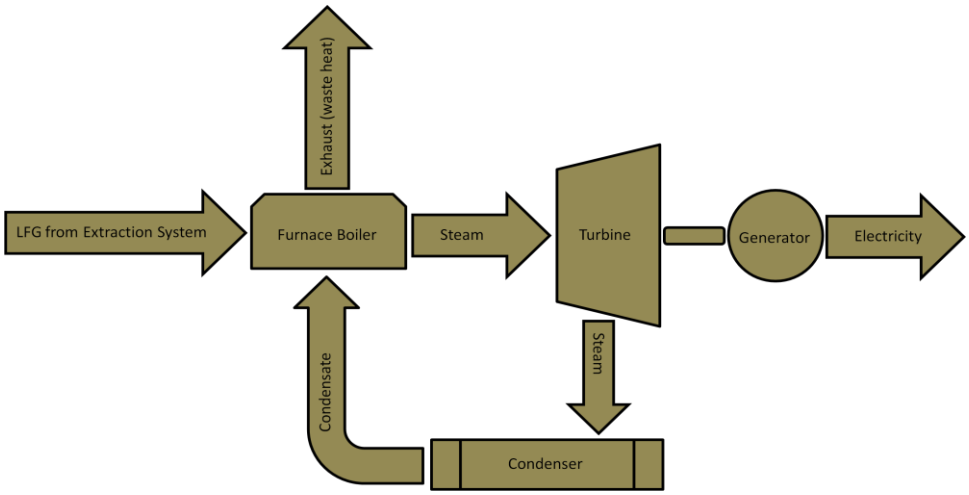


Source: Rajaram et al. (2012)

Steam Turbines

Steam turbines, or steam-Rankine systems, have been generating electricity since the mid-1800s. A heat source vaporizes a closed circuit water supply, which produces steam. The steam expands in the turbine and rotates the blades to generate electricity. The steam is then sent to a condenser where it is condensed back to water and returned to the feed stock (Rajaram et al. 2012). Figure 3 shows the schematic of a steam turbine engine using boiler heat as its heat supply.

Figure 3: Steam Turbine Schematic



Source: Rajaram et al. (2012)

Of the seven largest LFGTE systems in terms of electricity generating capacity, three systems run on steam turbines. The largest LFGTE project in the country is a 50 MW steam turbine operated by the Sanitation Districts of Los Angeles County (EPA 2013a).

Syngas

Recently, fuel additives are being studied to reduce emissions from LFG combustion (Kohn et al. 2011). Syngas, which is an additive blend of H_2 and CO, has been shown in one study to significantly reduce CO, unburned hydrocarbons (UHC), and NOx emissions. When mixed with syngas, LFG becomes more reactive which allows for a more complete combustion and reduction in exhaust emissions. In the study, LFG for an IC engine, a gas turbine, and a microturbine was mixed with syngas for 5%, 10%, and 15% blends. After combustion with LFG-syngas blends, emissions measurements were collected.

The syngas blend most efficient in terms of fuel conversion and focused effluent compounds was a 5% blend. At the 5% blend, CO emissions decreased from 802 ppm to 214 ppm, UHC emissions were cut by an order of magnitude from 113 ppm to 12 ppm and NOx concentrations dropped from 100 ppm to 62 ppm (Kohn et al. 2011). Though these concentrations are significantly less because of syngas, they are still two to three

times the permissible exposure limit set by the Occupational Safety and Health Administration.

With hydrogen, the additive created a more reactive fuel, so as the blend percentage increased, the flame temperature increased. As a result, higher concentrations of NOx formed with the blends greater than at 5% because those higher blends triggered the flame temperature exceed 1,800 K. At these temperatures and greater, NOx generation increases exponentially. The difference between CO and UHC emissions from 5% and higher blends was marginal (Kohn et al. 2011). This additive is the type technology sought after in New Source Reviews in areas of non-attainment under the Clean Air Act.

Organic Rankine Cycle (ORC)

ORC power generation is similar to a steam turbine system with the exception that the ORC volatizes an organic fluid instead of water molecules (Bove and Lunghi 2006). OCRs are preferable to steam turbines when low temperatures characterize the heat supply because some organic fluids have lower boiling points than water. For example, pentane, a commonly used working liquid, has a boiling point of 36.1 ^{o}C. ORCs are commonly applied in geothermal energy production because ORCs accommodate the low available temperature range the subsurface provides compared to combustion (Hawkins et al. 2011).

Figure 4 shows an ORC schematic diagram using engine waste heat. The engine supplies waste heat and vaporizes the organic liquid which turns the turbine blades to power an electric generator. Combined cycle electricity generation brings these systems together to increase the electrical efficiency of the LFG. ORC design has self-contained VOCs from the organic liquid pass through a recuperator to a condenser which brings the VOC back from the vapor gas state to liquid state. The recuperator captures some of the turbine effluent vapor heat and transfers it to the condensed organic liquid for greater heating efficiencies. After the supply heat passes through the evaporator, it heats the

condensed organic liquid for a second time. Finally, the organic liquid reaches the evaporator and the process repeats itself.

Recently studies were focused on reducing the risk of fires from ORCs, especially when there are multiple waste heat sources. Gewald et al. (2012) examines the efficiencies of an ORC with a thermal oil heat exchanger as an intermediary between the supply heat and the organic fluid since most organic fluids are very flammable. The oil heat exchanger regulates the temperature the working fluid which reduces the chance of unintended fire; however, the heat exchanger compromises the ORC performance. The system-wide electrical efficiency decreased by 0.4% when pentane is used as the organic liquid and 1.3% when silicone liquid is used. This translates into a 0.2 to 0.7 MW capacity loss. Figure 5 shows the ORC design with added safety component.

Figure 4: Organic Rankine Cycle Schematic Installed

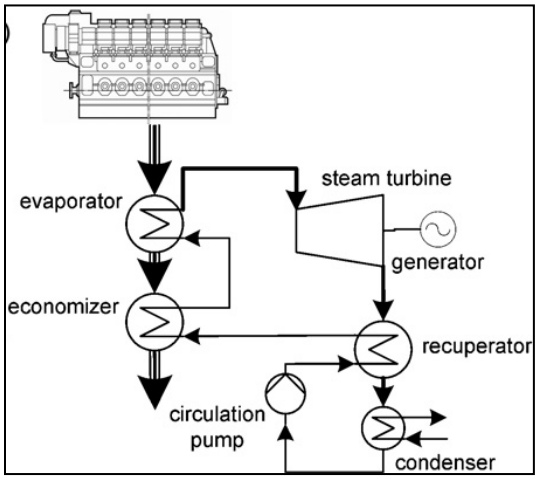
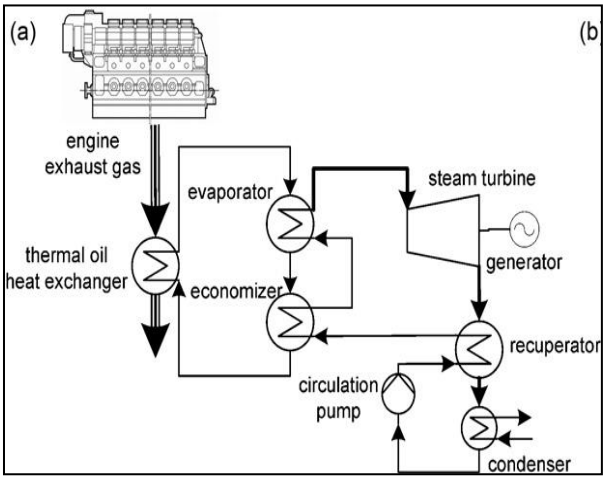


Figure 5: Organic Rankine Cycle with Safety Heat Exchanger



Source: Gewald et al. (2012)

According to the EPA's Landfill Methane Outreach Program, no ORCs currently use LFG as a heat source; however a small scale ORC engine operated for a period of time at a Danville, Illinois landfill (EPA 2013a). ORC underuse is attributed to the fact that steam turbines have higher heat transfer efficiency, so they create more power using the same LFG flows.

Stirling Cycle

In the early 1900s, the diesel engine surpassed the Stirling Cycle engine and the engine of choice due to greater efficiencies and power densities (Bove and Lunghi 2006). However, with increasing fossil fuel costs and supply of alternative fuels, the Stirling Cycle engine may be making a comeback. Like the ORC, low temperatures for heat exchange requirements make the Stirling Cycle engine more attractive than traditional power generators in certain settings. Landfills with low flow rates or low LFG quality are prime candidates for Stirling Cycle engines. As the Stirling Cycle is an external combustion process, fuel sources with high impurity concentrations like siloxanes may be used because internal engine parts are not exposed to LFG or its exhaust (Thombare and Verma 2008).

The general concept of a Stirling Cycle engine is to direct or indirect external combustion to heat a working fluid which drives a piston to generate electricity. A regenerative heat exchanger intermittently transfers heat from a source, like a LFG burner or waste heat, to expand and contract a working fluid. The working fluid moves a piston through a chamber which turns a spin wheel to rotate an electricity generator crankshaft. After the heated working fluid expands, it cools and flows to another chamber and moves a displacer piston, which is also attached to the generator crankshaft. The component, which allows heat exchangers to operate under low temperatures to pressurize the working fluid, is that the working fluid is usually helium. (Thombare and Verma 2008)

Stirling cycle engine application to LFGTE is still developing with only two small scale engines in operation in the US. Piston configuration design and alternative working fluids are a few of the current improvements being made to a system that was invented almost 200 years ago. This LFGTE alternative has the potential to service smaller landfills (<8M tons in place) as a secondary electricity source in a combined cycle system once more efficiencies are achieved (Roe et al. 1998).

Table 6 lists key characteristics for each of the LFGTE electricity generating alternatives. With LFG generation and transport explained and current LFGTE alternative presented, the last step in mitigating methane from landfill is treatment of fugitive emissions not captured by the LFG extraction system. This paper offers PMOB as an effective treatment option to mitigate fugitive emissions. The follow section explains PMOBs in detail.

Table 6: LFGTE Alternative Summary

LFGTE Electricity-Generating Alternative	Capacity Range (MW)	Flow Range (cfm)	Electric Efficiency	Capital/Annual O&M Costs per MW[c]
Cogeneration (CHP)	0.1-11[a]	50-4,200[a]	60%[d]	$2.3M/$210,000
Combined Cycle	2-23.5[a]	5,000-25,000[b]	47-50%[e]	Dependent on technology
Gas Turbine	1.5-14.7[a]	4,000-20,000[b]	28%[f]	$1.4M/$130,000
Microturbine	0.1-1.2[a], 0.03-0.25[c]	<100[b]	15-25%[b]	$5.5M/$380,000
Organic Rankine Cycle	0.2[a]	NA	18%[f]	$2.3M/$210,000
Internal Combustion Engine	0.1-14.9[a]	150-5,000[b]	33%[f]	$1.7M/$180,000
Steam Turbine	0.5-50[a]	6,000-25,000[b]	40-45%[b]	$1.4M/$130,000
Stirling Cycle	0.1-0.22[a]	NA	20-40%[b], 38.58%[f]	$2.3M/$210,000

[a]All values except noted from EPA 2013a; [b]Rajaram et al. 2012; [c]EPA 2011; [d]Staub et al. 2011; [e]Zeman 2010; [f] Bove and Lunghi 2006

Passive Methane Oxidation Biocovers

While LFG collection systems extract the majority of the LFG generated by landfills, landfills are still culpable for fugitive emission to the atmosphere. Depending on the LFG collection system and surface cover, fugitive emissions range from 5% to 65% of total methane generation that goes uncaptured by LFG extraction systems. US EPA AP-42 protocol sets a default value of 25% for fugitive emissions (Abichou et al. 2010). At 1,528 Mg CH_4 yr^{-1} of LFG generation at a typical one million-ton closed landfill (Lombardi et al. 2006), fugitive emissions constitute significant GHG emissions.

Within the last 25 years, researchers examined the impact of cover soils on fugitive LFG emissions (Whalen and Reeburgh 1990). It was observed that methane concentration in landfill soil gas decreased as it was transported through the cover surface. Bacteria found in the upper soil cover layer were oxidizing methane and converting it to carbon dioxide. Prior to Whalen and Reeburgh (1990), most of the methane oxidation research focused on marine environments (Harrtis, 1980; Lidstrom, 1984). With methane emissions at landfills identified as a significant source of anthropogenic GHG emissions, this biochemical process and the development of engineered PMOBs to exploit this process garnered more attention in the research community (Albanna and Fernandes 2009; Roncato and Cabral 2012; Abdolahzadeh et al. 2010; Huber-Humer et al. 2008). PMOB have been proven in laboratory-controlled environments to promote enough methane oxidation in the substrate layer to reduce methane emissions by ~99% (Roncato and Cabral 2012; Chanton et al. 2011). Supporting these findings are field-scale experiments, which show comparable methane removal efficiencies under high source fluxes (Abichou et al. 2010).

Most research since Whalen and Reeburgh (1990) has focused on optimizing methane oxidation rates by altering site specific parameters like soil temperature or moisture content (Albanna and Fernandes 2009). Although no field-scale PMOB exist (Abichou, 2013), successful laboratory-controlled experiments were replicated in pilot-scale field

studies (Cabral et al. 2010). This section explains the bacteria-driven methane oxidation process, design components of a PMOB, key site-specific parameters, and how the methane oxidation rate is calculated.

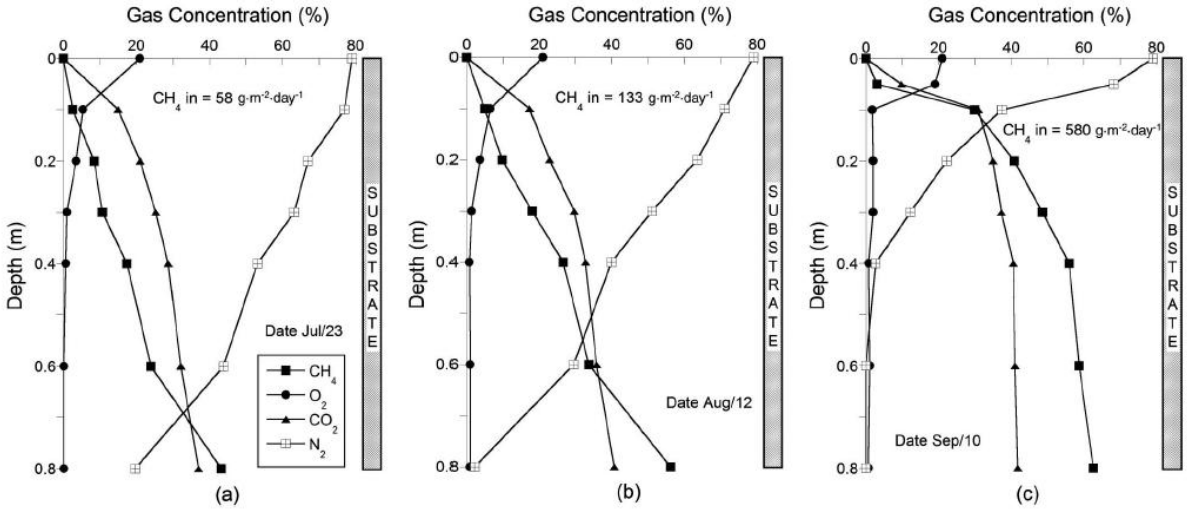
The Methane Oxidation Process

Methane oxidation is a biochemical reaction, which occurs in the uppermost layer of a landfill cover. Vertically migrating LFG, containing approximately 50% methane, come in contact with bacteria, known as methanotrophs, and cause an aerobic oxidizing reaction. The methanotrophs oxidize the methane over a series of sub reactions catalyzed by enzymes to form CO_2 from about 1 meter below ground surface to ground surface. A simplified exothermic reaction (Chiemchaisri et al. 2012) is given by:

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O + 210.8 \text{ kCal mol}^{-1} \quad \text{(Rxn 6)}$$

In a column experiment with two different PMOB designs, the measured CO_2 and methane percent concentrations by volume--of which methane was initially greater--converged in the substrate layer. The CO_2 share continued to reduce at a slower rate than methane as atmospheric nitrogen and oxygen became the dominant elements in the LFG. Across the substrate layer (0 m to 0.8 m), the percent by volume oxygen and nitrogen in the LFG equilibrate with the atmosphere and transition the soil from anaerobic to aerobic (Roncato and Cabral 2012). Oxygen supply is an important parameter in methanotroph proliferation. Figure 6 shows the substrate layer oxidation results using three different influent LFG flow rates. The source flow rate is one of the variables, which determine the CO_2 production rate and the methane removal rate.

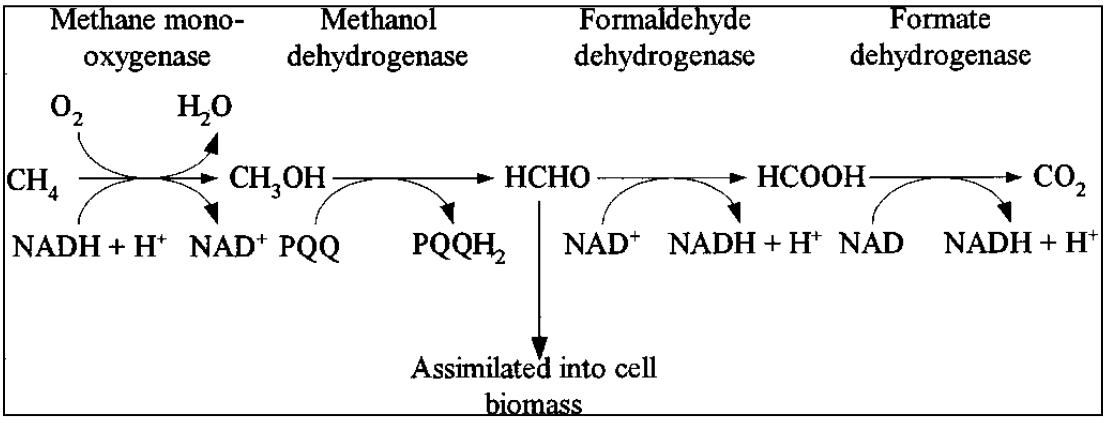
Figure 6: LFG Composition through the Substrate Layer



Source: Roncato and Cabral (2012)

A more detailed reaction sequence with the methanotroph-generated enzymes is presented in Figure 7. Organic compounds formed during the oxidation process include methanol, formaldehyde, cellular carbon, and formic acid. Mono-oxygenase enzymes from methanotrophs are coupled with a reductant, nicotinamide adenine dinucleotide (NADH), to hydroxylate the methane to methanol (CH_3OH).

Figure 7: Methane Oxidation Subreactions



Source: Murrell (1994)

Then, dehydrogenase enzymes, coupled with pyrrolo-quinoline quinone (PQQ), oxidized methanol to formaldehyde (HCHO). When formaldehyde is produced, not only does it oxidize into formic acid (HCOOH) with the aid of another methanotrophic enzyme and

NADH, but it is converted to cellular material through pathways from two types of methanotrophic bacteria. Some methanotrophs skip formic acid production and proceed directly to CO_2 generation when the formate dehydrogenase enzyme is limited. Otherwise, formate dehydrogenase enzymes and NAD oxidize the formic acid into CO_2 (Anthony 1982). If the right conditions are present with adequate oxygen and enzyme-generating methanotrophs, this oxidation reaction is self-sustaining.

Landfill cover design to promote methanotrophic-driven methane oxidation

Traditional landfills use daily soil surface covers while closed landfills are required by Resource Conservation and Recovery Act (RCRA) Subtitle C or D to apply more complex barrier covers to prevent water intrusion[4] and allow LFG extraction, when applicable. Figures 8 and 9 present a cross section of two RCRA covers: a Subtitle C cover for hazardous waste landfills and a Subtitle D cover for nonhazardous waste landfills. RCRA requires hazardous waste landfills to install LFG extraction while nonhazardous waste landfills are allowed to simply compact a soil layer above the waste without a LFG extraction system (Hauser 2009).

[4] Maximum saturated hydraulic conductivity is $1X10^{-5}$ cm s^{-1}.

Figure 8: RCRA Subtitle C Cover

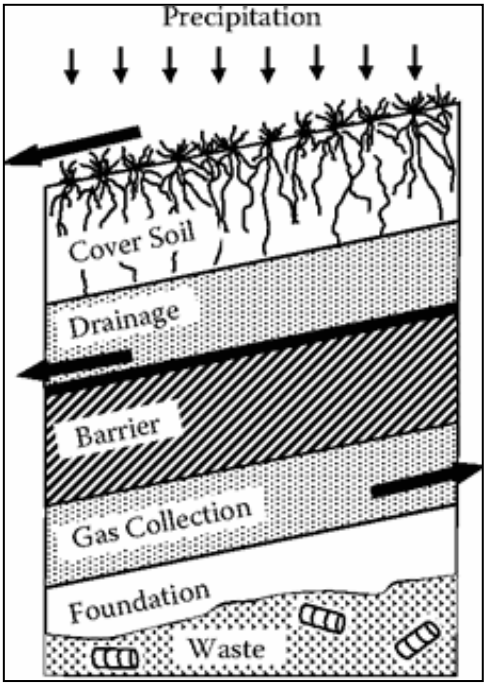
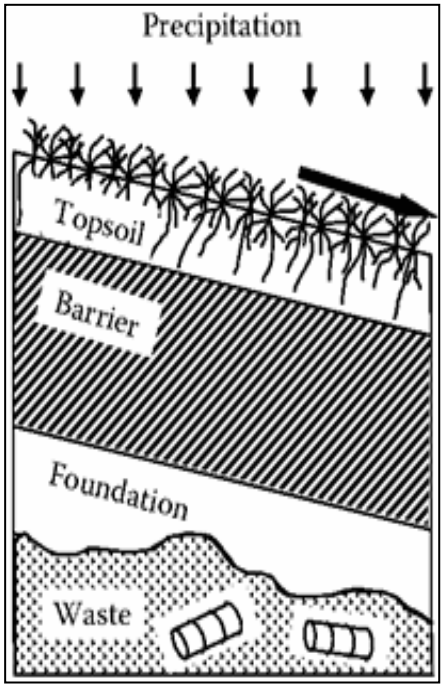


Figure 9: RCRA Subtitle D Cover



Source: Hauser 2009

This report does not as much suggest deviating from the RCRA design, but enhancing it. PMOBs can optimize methane oxidation in the upper layers of a conventional RCRA landfill cover because PMOBs are specifically designed to promote methanotrophic-driven aerobic methane oxidation. PMOBs are the same as the upper three layers of the Subtitle C cover: a geomembrane liner (barrier), a gas distribution layer (GDL) (also utilized for drainage), a substrate layer with a vegetative surface layer (cover soil). The only deviation is that the porosity in the substrate layer is higher than that of a Subtitle C cover to create a favorable environment for methanotrophs.

Geomembrane Liner: In traditional landfill covers, geomembrane liners, usually made of 1-mm thick high density polyethelene, cover the waste to prevent water infiltration creating unnecessary leachate, and to some degree, contain methane migration to beneath ground surface. This layer is the most impermeable layer in the entire landfill cover. However, geomembrane liners for PMOB have another use. It protects the gas distribution layer from upward migration of moisture (Cabral et al. 2010). Excess

moisture in the GDL, as described below, prevents the GDL from performing its intended purpose of equilibrating LFG distribution.

Gas Distribution Layer: As with other designed soil covers, GDLs are constructed to prevent concentrated flux zones which put undue stress on the substrate layer. This can create oxidation inefficiencies. Some PMOB experiments use relatively thin GDLs (Rose et al. 2008) with thicknesses around 0.2 m. According to Roncato and Cabral (2012), smaller oxidation efficiency fluctuations occurred in field experiments with a thicker GDL (0.8 m) than a thinner GDL (0.3 m) because concentration gradients had a greater distance to equilibrate before they reached the substrate layer.

Substrate Layer: Because methanotrophs require oxygen, moisture, and a steady supply of methane, most experiments utilize highly organic media for the substrate layer such as garden and yard waste (Abichou et al. 2006) or compost (Barlaz et al. 2004). Over a three year study of various substrate layer media,[5] Huber-Humer et al. (2008) found 1.2 m of compost was the most efficient design for a PMOB. The study concluded that the thicker the substrate layer, the longer the resident time will be for oxygen to infiltrate the layer and oxidation to occur. Based on a literature review, Table 7 presents the optimal substrate layer depth for methane to oxidize. All depths are within the 1.2 m depth proposed by Huber-Humer et al. (2008).

Table 7: Optimal methane oxidation depth

Report	Depth from ground surface (cm)
Abichou et al. (2006)	15-30[a]; 15-70[b]
Czepiel et al. (1996)	5-10
Visvanathan et al. (1999)	15-40
Whalen and Reeburgh (1990)	3-12
Nozhevnikova et al. (1993)	40-60
Borjesson and Svensson (1997)	40-60
Kightley et al. (1995)	20-30
Humer and Lechner (2001)	40-90[c]

[a]Sites 2 and 3; [b]Site 1; [c]Cover composed of sewage sludge compost and MSW compost; Source: Abichou et al. (2006)

[5] Manure, well-structured compost, waste substrates

When designing a PMOB the pore size differential between the substrate layer and the GDL should be taken into account. This differential impacts the moisture content in the substrate layer which influences the oxidation rate. GDLs are characterized with coarser-grained material than the substrate layer. As a result, capillary forces in substrate layer transport moisture from the GDL to the substrate layer, so equilibrium moisture contents between the two layers are skewed to the substrate layer (Abdolahzadeh et al. 2010).

Surface Vegetation: Fully developed surface vegetation is important to a productive PMOB. Rocanto et al. (2012) concluded that the reason water saturation was higher in their column experiments than in the field, which resulted in lower oxidation rates, was because vegetation was not available to transpire water molecules. Additionally, nutrients from surface vegetation promote microbial activity in the soil, which create a more favorable oxidation environment for the methanotrophs (Cabral et al. 2010).

Calculating Methane Oxidation in Passive Biocovers

The process of accurately calculating PMOB productivity is progressing. Researchers are gathering empirical information on the impacts of certain environmental conditions on the productivity of a PMOB (Chiemchaisri et al. 2012). During a 2010 computer-model simulation, Yuan and Abichou (2010) reported that it would take 22 years before a PMOB achieved 100% removal of fugitive emissions under current designs, so opportunities for optimization exist.

Conditions Impacting Oxidation Efficiency

Varying the PMOB design to exploit optimal methane oxidation conditions was the goal of most studies encountered in the literature review (Abichou and Chanton 2009) such as substrate thickness, temperature, porosity, oxygen concentration, and methane flux. These factors are presented in detail below. However, less researched environmental factors include VOC concentrations in LFG, soil type, and soil nutrients (Chiemchaisri et al. 2012) should be researched further.

Substrate thickness: Roncato and Cabral (2012) found that the biocover thickness does make a difference in the oxidation rates. With a 50% increase in thickness from 0.3 m to 0.45 m in a series of column experiments, the maximum methane oxidation rate increased from 75 g CH_4/m^2d to 141 g CH_4/m^2d. Both columns had similar water saturation, so LFG transport was comparable.

In a 2010 study, methanotrophs were recorded at even concentrations across the substrate layer (0.8 m). This surprised researchers, who expected the concentration of methanotrophs in the substrate layer to be the greatest in the highest temperature and most moist soil zones of the PMOB (Cabral et al. 2010). As a result, substrate layer thickness does matter. The thicker the layer, the more methanotrophs are present to oxidize methane at a faster rate (Maciel and Juca 2000).

Temperature: Because methane oxidation is an exothermic reaction, biocover layers are naturally warmer than air temperature. Three different studies conclude that the optimal temperature for maximum oxidation efficiency is 33^oC (Roncato and Cabral 2012; Humer and Lechner 1999; Scheutz et al. 2009). Hanson et al. (2010) reports optimal oxidation conditions at soil temperatures between 20 and 30 degrees Celsius. Below freezing, air temperatures still have the capacity to freeze the ground and halt methane oxidation reactions. Soil temperatures over 50^oC create an inhabitable environment for the methanotrophs, and methane oxidation does not take place (Zeiss 2006). Because methane production occurs in the insulated center of a landfill cell, landfills produce methane year round. This unfortunately means the highest methane emissions released from a landfill with a PMOB occur in winter. Roncato and Cabral (2012) observed this phenomenon one fall when a cold spell dropped the average biocover layer temperature from 28^oC to 14^oC. The methane oxidation efficiency in the PMOBs dropped from approximately 99% to 0% and 38%. Zeiss (2006) proposed a passive heating system to keep the substrate layer above 5^oC. Hansen et al. (2010) proposes an insulated cover to limit temperature variation. Table 8 lists studies which

measure the soil temperature and associated methane removal rate with various substrate layer compositions.

Table 8: Methane removal rates for certain substrate layer materials

Substrate Layer Material	Temperature (°C)	Methane Removal Rate (mg m^{-2} d^{-1})	Reference
Loamy landfill cover	25	48	Whalen and Reeburgh (1990)
Humic landfill cover	22	7.2	Jones and Nedwell (1993)
Sandy landfill cover	20	120-168	Kightley et al. (1995)
Sewage sludge and waste compost	18	24-384	Humer and Lechner (2000)
Precomposted yard or garden waste placed	25.7+/-8.8	48	Stern et al. (2007)
Composite sand/compost	12	804	Cabral et al. (2010)

Source: Cabral et al. (2010)

Porosity (Air-Filled and Water Saturation): Porosity is the void fraction of a unit mass. Porosity is either air-filled porosity[6] - or water saturation[7] (soil moisture). It is in these soil particle voids where LFG transport and reactions occur. The rate at which oxidation occurs in the gas phase is ten thousand times faster than in the aqueous phase, so air-filled porosity is a more important parameter (Whalen and Reeburgh 1990). Researchers show that the optimal water saturation for substrate layers is 40% to 80%. At water saturation greater than 80%, airways become constricted and diffusion is impeded (Roncato et al. 2010). Jung et al. (2009) proved diffusion is the primary gas transport mechanism in landfills. In addition to its impacts on LFG transport, excess water availability impacts the productivity of the methanotrophs. Heightened soil moisture reduced the oxygen-laden air filled porosity needed for aerobic methanotroph survival (Chiemchaisri et al. 2012). Not enough water also compromises methanotroph survival. At water saturation less than 13%, methanotrophs become inactive (Roncato et al. 2010).

[6] void fraction occupied by air molecules
[7] void fraction occupied by water molecules

A sensitivity analysis as part of Whalen and Reeburgh's 1990 landmark landfill study included a summary of methane oxidation rates with varying moisture content from 11% to 71% at 15% intervals. The analysis found that over a 12-hour period the methane concentration in a 70-gram landfill cover sample with optimal moisture content of 11% contained 35% less methane by concentration than a similar waterlogged sample with 71% moisture content. No noticeable deviation in methane concentration was observed until the moisture content was less than 40% (Whalen and Reeburgh 1990). Samples with moisture contents less than 11% also reduced the oxidation capacity. These results were confirmed six years later by another Whalen study on boreal soils (Whalen et al. 1996).

O_2 concentration: Because methanotrophs exist under aerobic conditions, oxygen is necessary for methane oxidation. As presented in Reaction 6, the molar ratio of methane to oxygen is two to one. One study by Kjeldsen et al. (1997) confirms the stoichiometics by reporting an optimal O_2/CH_4 ratio of 3.5-4.0g O_2/CH_4 or 1.75-2 moles O_2/CH_4. Chiemchaisri et al. (2001) reported maximum methane oxidation rates at a 6.5 g O_2/CH_4. At ratios greater than 6.5, the oxidation rate diminished. This is because as more methane molecules occupy the pore space, the fewer oxygen molecules are able to permeate through the PMOB to provide aerobic conditions for the methanotrophs (Yuan and Abichou 2010).

Methane Flux: Abichou et al. (2006) conducted a methane oxidation study at four unique sites without PMOBs at a landfill in Florida. The goal of the study was to established baseline oxidation rates prior to implementing engineering controls.[8] The researchers used the ^{13}C stable isotope technique to calculate those rates. The study found the methane flux reduction varied significantly across the 220 samples (-13.6 to

[8] In the 1997 IPCC and the US EPA set a default oxidation value for landfill sites without PMOB of 0%-10%. Since then research has concluded that the rate is much higher. The average oxidation rate from 42 studies is 36.4% with ambient temperature as a determining factor (Chanton et al. 2010)

1,755 g m^{-2} d^{-1}) collected. Most methane flux peaks shortly after landfills close and then steadily diminish over time until all the waste has biodegraded. Despite the flux variation, the study determined that there was no significant difference of oxidation rates among the four sites. A sensitivity analysis did establish an inverse correlation between methane flux through the cells and percent methane oxidation (Abichou et al. 2006). One likely reason is the residence time for the methanotrophs to consume methane is reduced with the higher flux. Another hypothesis is that the higher fluxes did not allow for adequate oxygen permeation which creates favorable aerobic conditions for the methanotrophs.

Equations for methane oxidation calculations

The simplest method for measuring and calculating methane oxidation through a PMOB in the field is with a chamber experiment. Once the methane flux rate (*J*) is calculated, the total fugitive emissions are subtracted from the surface emissions to calculate the PMOB methane removal rate (R_{CH4}). The following equation presents the methane oxidation rate:

$$R_{CH4} = J - (V/A * \frac{dC}{dt} * \varepsilon)$$

(Eqn 15)

where *J* (g m^{-2} d^{-1}) is the methane generation source flux from waste decomposition to the PMOB which accounts for LFG dispersion and advection (Eqn 11); *V* is the chamber volume (m^3); *A* is the landfill surface area covered by the chamber (m^2); and *dC/dt* is the time weighted average change in concentration (mg m^{-3} h^{-1}) in the chamber. Because most samples return values in mg m^{-3} and collection intervals are usually in the order of hours, a conversion factor (ε=0.024) must be applied to the equation to align with the source term units. Source flux assumes equal methane generation distribution across the landfill based on results from Abichou et al. (2006). For a simplified approach which

does not take transport mechanisms into account, substitute *J* with Q_{source} from Eqn 1 and the area of the landfill.

If the landfill has a LFG extraction system installed, a methane removal coefficient (χ) will need to be applied to *J* or Q_{source}. LFG extraction systems generate negative pressure on the landfill, so when air is drawn into the cover the oxygen content increases which creates a more favorable environment for methane oxidation (Huber-Humer et al. 2008). Based on US EPA AP-42 protocol, the methane oxidation rate with an active LFG system is give as:

$$R_{CH4} = \left(J * (1 - \omega)\right) - (V/A * \frac{dC}{dt} * \varepsilon)$$

(Eqn 16)

A default value approved by the US EPA for ω is 0.75 where 75% of the methane generated is captured by a LFG extraction system (Abichou et al. 2010). These simplified formulas do not take into account cover type or precipitation rates which should be considered when calculating oxidation rate for PMOBs.

Using the methane oxidation rate, the stoichiometric equation for methane oxidation yields a carbon dioxide conversion rate (R_{CO2}) (Perera et al. 2002). This is the rate of carbon dioxide that is generated from methane oxidation. The carbon dioxide conversion rate is calculated with this equation:

$$R_{CO2} = 0.7 * R_{CH4}$$

(Eqn 17)

where 0.7 is the stoichiometric conversion coefficient between carbon dioxide and methane, and R_{CO2} is the carbon dioxide generation rate (g m^{-2} d^{-1}). Through soil column experiments, Stein et al. (2001) determined 0.8 as the conversion coefficient.

Stable isotope tracers technique for estimating methane oxidation

On a field scale, there are limited means to calculating LFG subsurface and surface flux. However, recent studies show that stable carbon isotopes ^{12}C and ^{13}C work well to determine the percent of methane that is oxidized through methanotrophic-driven methane oxidation. In the atmosphere, ^{12}C dominates the carbon content at 99% while ^{13}C is around 1%. The exact ambient air ratio of ^{13}C to ^{12}C has been measured (R_{amb}=0.01142). Methanotrophs favor uptake of the lighter isotope ^{12}C over ^{13}C which changes the ratio and allows for oxidation estimation. Once methane emissions from a chamber experiment over a point on the landfill are collected, the sample is analyzed for its ^{12}C and ^{13}C content ($^{12}C_{sample}$.and $^{13}C_{sample}$). Samples for percent carbon content may be also collected downwind of the site. Though, these samples must take into account ambient air dilution. With the sample and ambient ratios, the ^{13}C carbon isotopic composition is calculated. Then, with additional site specific parameters, the oxidation rate is calculated. The following equation estimates the carbon isotopic composition ($\delta^{13}C\ \%_{sample}$):

$$\delta^{13}C\ \%_{sample} = \left(\left(\frac{\frac{^{13}C\ _{sample}}{^{12}C\ _{sample}}}{R_{amb}} \right) - 1 \right) * 1000$$

(Eqn 18)

A decreasing negative value for $\delta^{13}C\ \%_{sample}$ means more oxidation is occurring. With $\delta^{13}C\ \%_{sample}$ calculated, the methane oxidation frequency (f_{ox}) is given by:

$$f_{ox} = (\delta^{13}C\ \%_{sample} - \delta^{13}C\ \%_{orig})/(1000 * (\alpha_{ox} - \alpha_{trans})$$

(Eqn 19)

where the signature $\delta\ ^{13}C\ \%_{orig}$ in the anoxic zone of a landfill is -55%;and α_{trans} and α_{ox} are the isotope fractionation factors associated with microbial methane oxidation and methane transport, respectively. The isotope fractionation transport factor is

dependent on whether advection (α_{trans}=1) or diffusion (α_{trans}>1) dominates LFG transport. Studies have found that advection is the dominant transport mechanism (Borjesson et al. 2007). Once the fraction oxidation is known, the methane oxidation rate (MOR) can be calculated using this formula:

$$MOR = f_{ox} * (\frac{J}{1 - f_{ox}})$$

(Eqn 20)

where *J* (g CH_4 m^{-2} hr^{-1}) is the methane transport flux from Eqn 11; and f_{ox} is the fraction of methane that was oxidized by the cover calculated in Eqn 19 (Bojornssen 2007). From the Cabral et al. (2010) literature survey, the average methane oxidation rate across a variety of soil types ranged from -7.2-804 g m^{-2} d^{-1}.

Methane Mitigation Calculation for Closed Municipal Landfills Employing LFGTE and PMOB

With the methane mitigation effectiveness of LFGTE and PMOB explained in the previous sections, cities should implement policies which support development of these technologies at their closed municipal landfills. The combination of these two technologies will significantly reduce their annual GHG emissions and provide opportunities for energy cost-savings or added revenues if treated LFG or electricity is sold on the open market. This section quantifies those reductions and energy-generating opportunities for closed landfill by calculating methane generation, available flows for LFGTE, and remaining fugitive emissions treated by methanotrophs for a hypothetical closed municipal landfill. After mitigation, the final amount of GHG emissions is compared to emissions without mitigation alternatives. The hypothetical landfill under analysis has a combination of site characteristics and specifications from by Lombardi et al. (2006) and Morgan and Yang (2001). The Lombardi et al. (2006) landfill is small compared to other landfills (Morgan and Yang 2001), so all values presented are smaller in scale than the average 8-million ton landfill.

Methane generation is calculated with EPA LANDGEM v3.02. Once the annual and cumulative methane generation rates are known, flux values for captured LFG from extraction wells are determined to established available flow for LFGTE. It is assumed the fugitive emissions transport to the surface and pass through a PMOB where the methane is oxidized into CO_2. The final CO_2e emissions values from the electricity generation and CO_2 from the PMOB combined with offsetting emissions from traditional electricity sources foregone by generating electricity onsite are compared to LFG emissions if no engineering controls were applied.

LFG Generation Calculations and Results

The hypothetical landfill holds one-million tons of wet MSW (Lombardi et al. 2006). The waste was accepted over a five-year period at 200,000 tons per year. Lombardi et al. (2006) provides MSW composition, so the Ritzkowski and Stegmann methane formation potential (Eqns 2 and 3) was applied. The methane formation potential was within the range of published literature, but less than the New Source Performance Standard default value. For the methane generation rate, the New Source Performance Standards default value was used because this value is usually derived for empirical data. There is nothing unusual about the waste composition so the default NMOC was also used. Finally, Lombardi et al. (2006) specified the methane content as 49.57%. These values were inputted into LANDGEM 3.02v (Eqn 1) for annual and cumulative methane generation. Figures 10 and 11 present LFG generation and flow rates over an 80-year period from LANDGEM.

Figure 11: Annual LFG Generation Rates

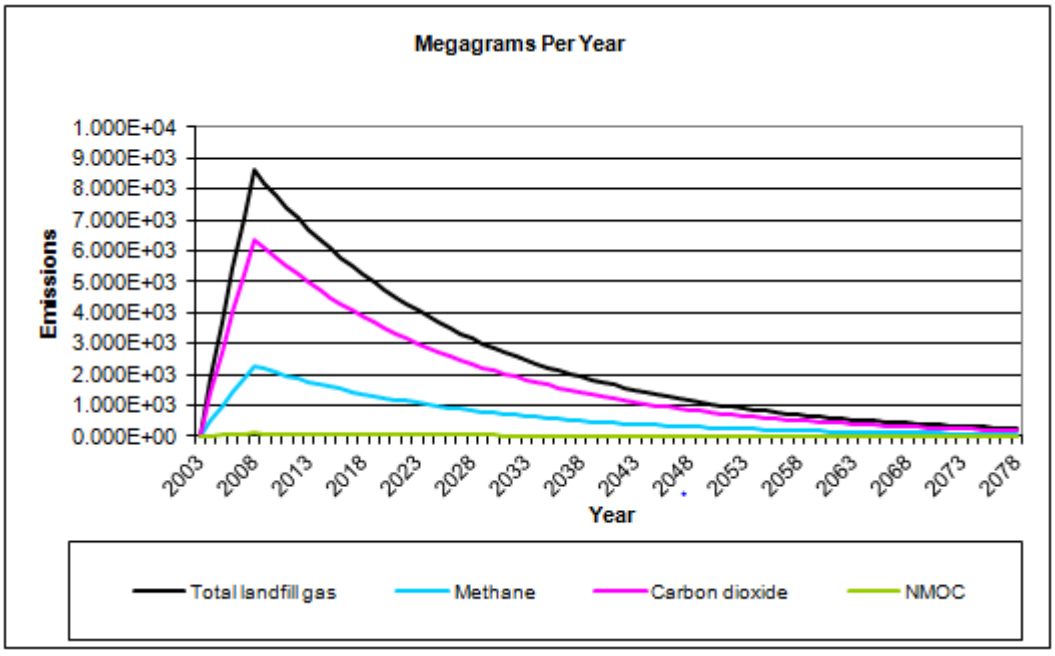
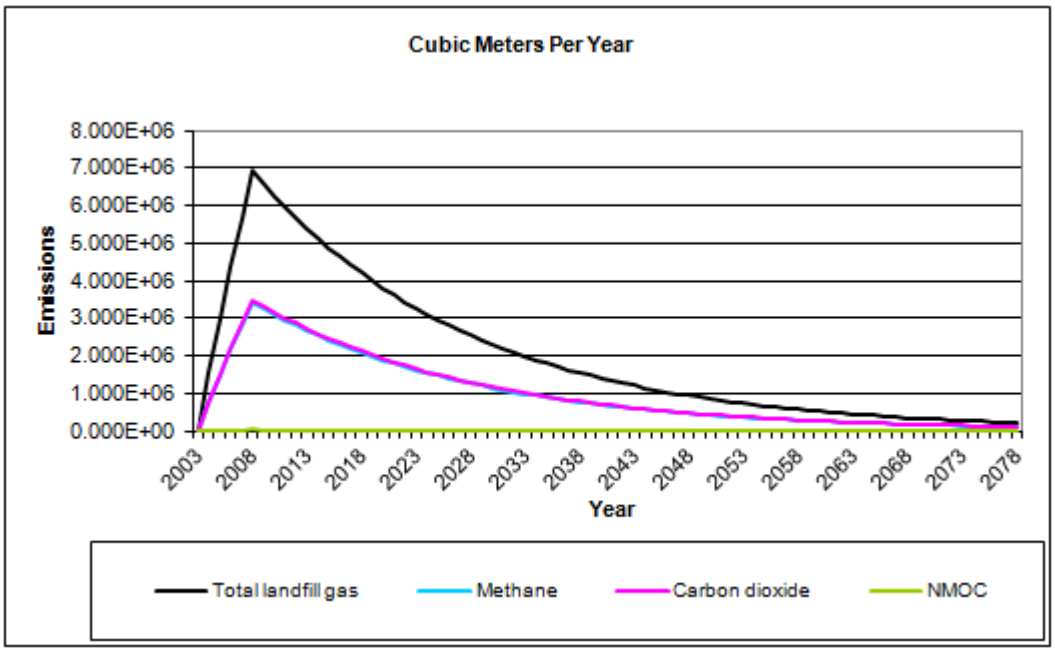


Figure 12: Annual LFG Flow Rates



These graphs show the importance of establishing controls during and immediately after the landfill is closed to capture a valuable fuel source at the height of its supply. Waiting to install a system even five years after closure may result in significant emissions and lost cost-savings from LFGTE operation. Cumulative methane and CO_2 emissions without controls over a 35 year period were summed. The total amount of CO_2e released from LFG generated during that period is 982,000 MT. Hundreds of landfills across the United States emit LFG into the atmosphere because they lack engineering controls necessary to mitigate emissions. This has been documented as one of the main contributors of anthropogenic methane sources in the atmosphere (EPA 2011).

Proposed LFGTE alternative with GHG reduction calculations

Now that the annual methane generation quantities are known, the landfill should be evaluated for LFGTE alternatives. Lombardi et al. (2006) provides LFG extraction system capture efficiencies over a 35-year period, so the available flux for LFGTE is easily calculated. In the first year of operation, LFG is not captured. In years two through five

while the landfill is active, the LFG capture efficiency is 50%. When the landfill closes, interim covers are applied. These seal the LFG in the landfill for a higher collection rate compared to daily, loose covers. Lombardi et al. (2006) assume the interim cover placed on the landfill after year 5 raises the collection efficiency to 80% for the next 30 years.

Assuming the generated LFG, which was captured by the LFG extraction system, was captured immediately and not influenced by transport forces, available methane flow rates ranged from 25.4 cfm in year 2 to 184 cfm in year 5 when the landfill closes and then down to 43 cfm in year 35. The most appropriate LFGTE alternate is a microturbine. Microturbines are not the most efficient LFGTE alternative in terms of their GHG reduction factor, but this site's alternatives are limited due to its available flow rates. Microturbines are designed to operate under low flow rates. Even for microturbines, though, these flows are on the lower end of their operating range, so electricity capacity will likely be less than 100 kW. At these rates large scale cogeneration is not an option. Burns & McDonnell (2012) evaluated microturbines with cogeneration for the Gundersen Health System and opted for an internal combustion engine because it could provide adequate heat for their facilities. Two LFGTE alternatives that could be pared with a microturbine in a combined cycle design is a Stirling Cycle engine due to its capability to handle low LFG flow rates or an Organic Rankine Cycle engine because of the low heating values for the working fluid. This combination should be examined further.

Microturbines have a methane destruction efficiency of 99.5%, meaning that almost all methane is combusted into CO_2. It can be assumed that a flare burned the LFG during landfill operations up to year 5. This would produce the same methane destruction results as a microturbine. From year 2 to year 35 the CO_2e emissions for carbon dioxide, either from LFG or combusted methane from the flare and microturbine is 90,000 MT CO_2e. Over the life of the inactive landfill, the extraction system collects the majority of the LFG and converts it into one-tenth its potential CO_2e mass.

Now consider the energy savings of a small microturbine. Capstone Turbine Corporation manufactures a 65 kW microturbine (Capstone 2013). If the turbine operated at 80% capacity 24 hours a day to power the LFG extraction pumps and treatment system, it would generate 455,000 kWh over the course of a year. The average price of a kWh in September 2013 according to the US IEA is 10.59 cents per kWh (EIA 2013g). This means the landfill would be saving almost $50,000 per year on electricity. Based on these savings and the capital and O&M costs of the microturbine found in Table 6, the payback time for the microturbine is around ten years.

The forgone emissions from electricity that would have originated from a conventional coal fired power plant should also be taken into consideration when weighing the advantages of LFGTE. The amount of CO_2e emitted from generating electricity from a coal-fired power plant is around 2.13 pounds per kWh. If a landfill needs 455,000 kWh per year, the resulting GHG emissions from their electricity needs are 440 MT CO_2e per year or 15,400 MT over 35 years.

LFG Transport and Passive Methane Oxidation Biocover Calculations and Results

From the LFG generation and collection calculations, fugitive emissions remain. Fugitive emissions are the influent flows for the PMOB. Fugitive emissions do experience transport force, so transport force calculations are used to determine the influent flux needed for PMOB calculations.

Assuming a homogenous clayey sand soil type, Eqn 11 was used to determine the methane and CO_2 mass flux from the cell through the lower cover to the PMOB. To calculate the diffusion coefficient for methane in soil, Eqn 7 was applied with soil porosity of 0.4 and volumetric air content at 40% which is within the range for optimal methane oxidation conditions. The concentration gradient was assumed uniform from LANDGEM source concentrations of 0.655 kg/m^3 for methane. The LFG flow velocity (Eqn 13) required LFG viscosity (Arigala et al. 1995), intrinsic permeability (Li et al. 2012),

and empirical pressure gradient values (Li et al. 2012). Estimated flow rate for methane was 0.014 g CH_4 m^{-2} s^{-1}. It is assumed all fugitive CO_2 passes through the PMOB unaltered.

Calculating the methane oxidation rate for a specific cover requires empirical data to establish a concentration gradient across the cover. In lieu of field measurements for this report, Chanton et al. (2011) plotted oxidation rates against flow rates for a variety of flows at a Florida landfill. At 0.014 g CH_4 m^{-2} s^{-1} using the linear regression, the oxidation efficiency is 10.4%. This low yield is due to the high flux. If the flux were lower, the oxidation efficiency would be much higher.

GHG Emissions Reductions from LFGTE and PMOB

Without controls, a one-million ton landfill may generate around 982,000 MT of CO_2e in its first 35 years. By installing a LFG extraction system and combusting the methane into CO_2 either through a flare or LFGTE, the landfill releases 90,000 MT of collected landfill gas. By using alternative energy, the landfill prevents, or is credited, 15,400 MT of CO_2e from off-setting conventional coal-fired power plant emissions. With a PMOB, the uncaptured GHG pass through an environment with active methanotrophs which reduce the CO_2e from 231,000 to 210,000 MT over 35 years. In total the GHG emissions are reduced by 68%.

Conclusions

Increased severity and frequency of precipitation events, prolonged droughts, and rising sea levels are a few of the many impacts of climate change. Events like Superstorm Sandy, which caused over $35 billion USD in damages to New Jersey and New York, show the effects of climate change that know no political or economic boundaries.

Due to the current political environment, the United States Congress has failed to act on comprehensive GHG pollution policy reform. Fortunately, GHG mitigation policy is not limited to federal directives. Coalitions, like the C40, show that some communities take climate change as a serious threat and as such, are willing to put forth policies to mitigate it.

Fortunately, these policies promote engineering controls, like LFGTE and PMOBs at municipal landfills. These controls have the potential to capture most of the methane generated by the landfills by converting it to either energy in the form of electricity or heat which a municipality could use or sell or CO_2, a less potent GHG. An added benefit to reducing emissions via LFGTE and PMOB is the direct human health and environmental risk reductions. Generating electricity from LFG offset potential mercury, SOx, NOx, and CO_2 emissions from coal-fired power plants and controls methane emissions which are an explosive hazard.

With the aid of LFGTE technologies, 1 million tons of municipal solid waste is capable of generating 0.8 MW of electricity (Rajaram et al. 2012). Most large US landfills contain 8 million tons of waste. In McKinney, Texas outside of Dallas/Ft. Worth, the North Texas Municipal Water District operates a 3.2 MW LFGTE plant at the former McKinney Landfill. In 2011, Raytheon purchased two-thirds of the site's electricity generating capabilities which provides 20 percent of their electricity needs throughout Raytheon sites in north Texas. As of December 2010, 24 LFGTE facilities were in operation in Texas (Wastebusinessjournal.com 2011).

The benefit of PMOBs versus other emerging landfill cap designs is that it is unnecessary for RCRA to be amended. RCRA Subtitle C does not specify the type of cover soil, so the PMOB construction of vegetation/substrate layer, permeable gas distribution layer, and geomembrane liner meets the design requirement for a RCRA Subtitle C landfill cover. Despite knowledge of microbial-driven methane oxidation for around 35 years, researchers have only begun to understand the methane oxidation efficiencies of PMOB. The IPCC and EPA established the industry standard for the oxidation rate from landfill covers at 0 to 10%; however, researchers have recorded oxidation rates as high as 100% (Cabral et al. 2010; Roncato and Cabral 2012). It was 25 years ago that the idea was applied to landfills, and a maximum oxidation rate has still not been established for this emerging technology.

This report's calculations show that LFGTE and PMOB will significantly reduce fugitive GHG emissions for an inactive landfill. In a hypothetical case study, a municipality with a one-ton landfill could potentially generate 455,000 kWh of electricity per year from a 65 kW microturbine while preventing 668,000 MT of CO_2e from entering the atmosphere. This policy would be comparable to erasing emissions generated from providing electricity for 2,625 homes or removing 3,741 cars from the road over the same 35-year period. With the consequences of climate change becoming ever more apparent and costly, municipalities like those in the C40 should continue to do their part by contributing to the solution.

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